The selection of protected areas in the face of fluctuating populations

Sarah F Jackson

Submitted for the degree of Doctor of Philosophy
Department of Animal and Plant Sciences
University of Sheffield

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SUMMARY

Protected areas are an integral part of the majority of national conservation strategies, and are seen by many as the most practical means of safeguarding biological diversity. Nonetheless, many such areas exist in name only or have been awarded protected status for their lack of economic potential rather than any genuine biological significance. Given the imminent extinction crisis it is essential that networks of protected areas are fully representative of key elements of biodiversity. To achieve this goal not only requires the efficient and effective selection of novel protected area networks, but also the regular evaluation of the current situation as regards existing/established networks, to identify and ultimately rectify possible weaknesses and gaps in coverage. Despite increasingly urgent calls for the development and application of comprehensive protected area evaluation systems, analyses of this type remain rare, particularly regarding the number of individuals sustained over time.

This thesis provides an in-depth analysis of the status of the existing network of wetland sites identified as nationally/internationally important for their wintering waterbird populations (Special Protection Areas and Ramsar Sites) in Great Britain. In addition, the aim is to examine the distribution patterns and population dynamics of selected waterbird species, and their implications for the selection and management of these wetland protected areas both in Great Britain and across the European Union; to examine the effectiveness and utility of alternative site selection methods, in particular the use of linear programming techniques, for real-world conservation issues; to provide suggestions for the improvement of the current network of protected areas both in Great Britain and across the European Union; and to provide recommendations and suggestions for waterbird conservation in general.

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DECLARATION

This thesis includes material from five papers (published, in press or submitted). Given that the thesis is partly formulated from these papers there is some inevitable duplication of content between some Chapters, although for clarity this has been kept to a minimum.

- Jackson, S. F., Gaston, K. J., and Kershaw, M. in press. Size matters: the value of small populations for wintering waterbirds. Animal Conservation. (Chapter 2)
- Jackson, S. F., Gaston, K. J., and Kershaw, M. in press. The performance of procedures for selecting conservation areas for waterbirds in Great Britain. Biological Conservation. (Chapter 4)
- Jackson, S. F., Kershaw, M., and Gaston, K. J. submitted. Identifying priority areas for waterbird conservation: data types and windows. Biological Conservation. (Chapter 5)
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All papers were based a joint idea by SFJ and KJG and SFJ conducted the analysis and drafted the manuscript.

CHAPTER 1: INTRODUCTION

"We might not be much good about protecting wildlife, but we know a lot about it"
(Marren, 2002)

UK Conservation:
people, policy and protected areas

1.1 Conservation potential

The UK is routinely dismissed in international conservation circles, although seldom is this disinterest explicitly recorded within the literature. Superficially, the overall lack of endemics, an almost entirely anthropogenic landscape, and a wealth of well-documented policy disasters suggest that it is little wonder. Over the last 50 years agricultural and industrial progress has seen: the blanket afforestation of uplands; motorways dissecting previously important sites for conservation (e.g. Hook Common Site of Special Scientific Interest (SSSI) in Hampshire and Aston Rowant National Nature Reserve (NNR)); new housing developments and out-of-town hypermarkets tearing up greenfield sites; the pollution of waterways from intensive agriculture and industry; the slaughter of countless numbers of migrating swans by electricity pylons and cables; the ploughing of ancient meadows; and the destruction of hedgerows to name but a few. Would it, therefore, be a catastrophic loss to global biodiversity if the UK were to sink without trace beneath the waves?

Paucity of endemics

Given the inevitable limitations on biodiversity conservation through competing land use, lack of resources and inadequate funding available for conservation purposes, it is perhaps unsurprising that endemism has become a central theme for conservation efforts (Myers et al. 2000, Conservation International 2003). Indeed, assigning priority to globally important areas aims to ensure maximum impact for conservation investment (Mittermeier et al. 1998, Myers et al. 2000). Following this rationale, Norman Myers published his now controversial set of 25 terrestrial hotspots, with the express purpose of providing a focus for global conservation investment (Myers et al. 2000). As none of the so-called 'hotspots' of biodiversity can be found in the UK is this confirmation that the value of UK diversity in global terms can largely be discounted?

The lack of endemic biodiversity in the UK is largely attributable to its glacial history and degree of isolation. Indeed, the elimination of the greater majority of previously occurring, temperate animals and plants took place during the period between the end of the ice-age and the severance of the land-bridge with Europe approximately 8,000-10,000years ago. As a result, the terrestrial and freshwater biodiversity of the UK is generally lower for many taxonomic groups than equivalent areas. Put into perspective, the forested area of the Philippines, which is roughly the same area as the UK (244,880km^2), supports approximately 5,832 endemic plant species and 518 non-fish vertebrates compared with only 16 plant and one vertebrate endemic found in the UK (Myers 2003). Additionally, the Solomon Islands, in an area of approximately 29,790km^2, considerably less than that of the UK, supports 21 mammalian and 43 avian endemics. Whereas, Cuba, in an area less than half that of the UK (114,525km^2) supports 3229 endemic species of plant (Groombridge and Jenkins 2002).

Agricultural legacy

In addition to the historical consequences of glaciations, enduring UK biodiversity has been further constrained and eroded by the direct and indirect intervention of humans. So much so, that the modern-day landscape of the UK is almost entirely anthropogenic (Marren 2002). However, despite a diverse range of adverse factors, the most significant influence, both historically and currently is that of agriculture. In fact, approximately 77% of the UK land area is controlled by agricultural regimes (DEFRA 2003) and sheep or deer graze even our remaining remote areas.

Historically, agriculture has shaped many European landscapes, mainly as a result of generous support policies. This has given rise to semi-natural environments, solely dependent on the continuation of farming. Indeed, European farmers directly manage and maintain approximately 44% of land as utilisable area, and when the remaining land they own or rent is taken into account, farmers effectively control more than half the total land area of Europe (Ostermann 1998, Pain and Donald 2002). There is, however, large variation in the extent of these agricultural areas across Europe: less than 10% in Sweden and Finland, 64% in Denmark and 81% in the Republic of Ireland (Ostermann 1998). Encouraged by various Common Agricultural Policy (CAP) incentives, advances in farming technology have had marked effects on UK biodiversity. In particular, the polarisation of agricultural activities from low-intensity systems either to intensification or to abandonment has reduced the value of agricultural lands for biodiversity (Ostermann 1998). Detrimental effects caused by this polarisation include, reduced soil quality, increased erosion and soil compaction, alterations to water flow, increased run off and pollution of ground water from chemical applications, draining of wet pastures, destruction of hedgerows and the loss of farmland birds and agricultural 'weeds' such as the marigold (*Tagetes spp.*). Nonetheless, some agri-environmental schemes, aimed at the maintenance or establishment of low intensity farming systems, were integrated into the CAP, following its reform in 1992 (Directive 2078/92/EEC). Unfortunately, many of the payments made to farmers to promote conservation have been unable to compete with those offered by the government for agricultural purposes. For example, in the Cambrian mountains a farmer is paid £22 per acre to encourage heather moorland, but £30 per head for sheep (Marren 2002). Historically, therefore, conservation activities in the UK have come a poor second to agricultural development.

Table 1.1: The numbers and area of statutory and non-statutory protected areas at 31st March 2002 (DERFA 2003). [1]Some areas may be included in more than one category; [2]Great Britain only; [3]Northern Ireland; [4]Figure excludes sites classified in dependent territories; [5]Scotland only. Statutory protected areas are afforded strict protection through strong legal and institutional structures, whilst non-statutory protected areas represent varying degrees of protection, institutional capacity and defensibility that are consistently weaker than statutory conservation areas.

Status[1]	Number	Area (10^3ha)
Statutory		
National Nature Reserve	395	245
Local Nature Reserve[2]	780	45
Sites of Special Scientific Interest[2]	6565	2286
Areas of Special Scientific Interest[3]	196	92
Marine Nature Reserves	3	21
Special Protection Areas	234	1312
Candidate Special Areas of Conservation	567	2359
Ramsar sites[4]	144	759
Environmentally Sensitive Areas	43	3190
Area of Outstanding Natural Beauty	50	2407
National Scenic Areas[5]	40	1002
Non-statutory		
Biosphere Reserves	13	44
Biogenetic Reserves	18	8

In situ conservation measures

Nonetheless, despite progressive fragmentation and degradation, on the face of it, biodiversity in the UK appears extremely well accounted for and protected. Indeed, there are at least 16 different statutory categories of protected area within the UK, all of which provide some level of protection for wildlife and habitats (Table 1.1). Some of these categories originate from global or pan-European Union (EU) incentives, including the World Heritage Convention, the Ramsar Convention on Wetlands of International Importance, the UNESCO Biosphere Conference and the Bern Convention. It has even been postulated that 'Nature reserves are the greatest achievement of half a century of nature conservation in Britain' (Marren 2002).

However, as Peter Marren points out in his recent book '*Nature Conservation*', much contemporary conservation policy/law has been arrived at in an *ad hoc* manner, resulting in many anomalies and numerous disasters for wildlife conservation (Marren 2002). Many such mistakes can, however, be largely attributed to conflicts of interest between agriculture and conservation, combined with a parochial attitude to rarity and threat. Indeed, this insular view of conservation priority has seen vast sums of money effectively wasted on species which are locally rare, but globally numerous (Peel and Speight 1995). One particular example is that of an English nature reserve, which is under management intended to encourage blue butterflies, at the expense of one of the country's largest populations of Crested Newts (*Triturus cristatus*). Ironically, blue butterflies are abundant across Europe, whilst crested newts are in decline even in their last substantial European population (Peel and Speight 1995).

1.2 Redeeming features

Despite the limitations imposed by its glacial and post-glacial history and the changes wrought by human occupation, the UK makes an important contribution to regional and global biodiversity. While its estimated total of some 130,000 species is not large in global terms (Table 1.2), it retains important biological communities characteristic of humid, and semi-humid, temperate climates and supports important marine and coastal populations of flora and fauna.

Arguably one of the most significant contributions to global biodiversity is that of the sheer number and diversity of migratory waterbirds common to wetlands during the winter period. Broadly, the wetlands of Northwest Europe regularly support between 11 and 13 million waterbirds (Gilissen et al. 2002) and for certain species, counts taken from Wetlands International's latest International Waterbird Census exceeded 1 million individuals (European white-fronted goose, wigeon (*Anas penelope*), mallard (*Anas platyrhynchos*), coot (*Fulica atra*), northern lapwing (*Vanellus vanellus*), dunlin (*Calidris alpina*), black-headed gull (*Larus ridibundus*)). The degradation and loss of suitable habitat for these species in the UK will ultimately result in a loss to global biodiversity, particularly at the population and individual levels.

The international importance of the UK for wintering waterbirds can be largely attributable to two factors: a) its geographic location and b) the diversity and extent of wetland habitats. First, the UK lies at the intersection of migratory pathways for individuals from arctic

Canada, Greenland, Iceland, northern Europe and Siberia (Stroud et al. 1990, 2001). As a result, the coastal and inland waters of the UK regularly support in excess of three million individuals (Pollitt et al. 2003). Many of these winter in dense aggregations and, therefore, critically depend on wetlands for feeding and roosting.

Table 1.2: The numbers of terrestrial and freshwater species in the UK compared with recent global estimates of described species in major groups. Adapted from http://www.chm.org.uk/library/ukprofilch.htm).

Group	British species	World species
Viruses	>1,600	>5,000
Bacteria	>1,600	>4,000
Protozoa	>20,000	>40,000
Algae	>20,000	>40,000
Fungi	>15,000	>70,000
Ferns	80	>12,000
Bryophytes	1,000	>14,000
Lichens	1,500	>7,000
Flowering plants	1,400	>250,000
Non-arthropod invertebrates	>3,000	>90,000
Insects	22,500	>1,000,000
Arthropods other than insects	>3,000	>190,000
Freshwater fish	38	>8,500
Amphibians	6	>4,000
Reptiles	6	>6,500
Breeding birds	210	9,881
Wintering birds	180	-
Mammals	48	4,327
TOTAL	>90,600	>1,770,000

For example, although the winter distribution of whooper swan (*Cygnus cygnus*) stretches from Iceland, the UK, Scandinavia, Russia and Siberia to Poland, the Caspian Sea, Turkestan and China, approximately 42% of individuals are supported by UK wetlands (see Chapter 2). Similarly, the winter range for Eurasian wigeon (*Anas penelope*) extends from Iceland, the UK and Scandinavia, across Russia and Siberia to Poland, eastern Germany and China. However, approximately 29% of this biogeographic population overwinters in the UK. As a final example, the winter range for common teal (*Anas crecca*) extends from Iceland, the UK and northern Scandinavia, across Russia and Siberia, France, Italy, Turkey, China, Mongolia, Japan, the Aleutian islands and Alaska. Nonetheless, approximately 33% of the population winters in the UK.

In addition to the provision of feeding and roosting areas, for other species the UK is a stop-over, on the way to or from wintering/breeding grounds in Africa. For these individuals, wetlands in the UK provide vital staging posts and an invaluable refuge during periods of bad weather. Migratory individuals critically rely on these staging areas for the provision of supplementary 'fuel' to complete their journey. Specifically, studies on lesser snow geese (*Anser caerulescens*) in North America have shown that clutch sizes are positively correlated with food availability on staging areas (Ankney and MacInnes 1978, Davies and Cooke 1983, Krapu and Reinecke 1992).

The diversity of wetlands can be largely attributed to daily, seasonal or longer-term hydrological fluctuations (Gibbs 1995, 2000, Hailes 1996). These factors produce a wide range of wetland types, which support an exceptionally diverse range of species. In fact, many rare and endangered plant and animal species are wetland obligates for some or all of their life span (Hartig et al. 1997). Further, the greater proportion of all Red Data Book invertebrates are critically dependent on wetlands for their survival (Peel and Speight 1995). Wetlands are, however, at risk from a wide variety of factors. Not least, from drainage for agriculture, settlement and urbanisation, pollution and hunting (Guillemain et al. 2002, Stuip et al. 2002); various governments, including those of the USA and the UK, have historically encouraged the drainage of wetlands to increase agricultural yields (Wheeler et al. 1995). Such policy decisions, in conjunction with increasing population densities in wetland catchments, and intensive landuse pressures mean that the European region is considered one of the most complex and challenging to be served by the Ramsar Bureau (Ramsar Convention Bureau 2002b).

1.3 Waterbird conservation

Waterbird conservation is necessary for a combination of moral, selfish and legal reasons. First, there is a moral argument, which suggests that we should aim to ensure that future generations enjoy the same quality of life as we currently experience. At the most basic level it could, therefore, be argued that we have a moral obligation to ensure that these species persist to ensure inter-generational equity. In other words we should aim towards: 'Meeting the needs of the present without compromising the ability of future generations to meet their own needs' (WCED 1987).

Second, waterbirds ultimately will impact on the lives of even those who hold no regard for wildlife or conservation. Birds have widespread popular appeal and therefore make good flagship species for mobilising volunteer-based monitoring networks, as well as for education and advocacy within civil society. Indeed, the appeal of birds in the UK is exemplified by membership figures for the Royal Society for the Protection of Birds (RSPB). Currently there are more members than that of any political party, which equates to an estimated 4000 members for every species of breeding bird in the UK (Marren 2002). Indeed, their long migrations and tendency to concentrate in large numbers mean these birds are highly valued by many stakeholders, including local human populations, tourists, associated enterprises, hunters (sport and subsistence), bird watchers and researchers (Kushlan et al. 2002). Throughout history, waterbirds figured prominently in human culture, serving as sources of food and ornamentation. For example, in the 1800s and early 1900s it was considered highly fashionable to adorn hats with feathers from species such as the snowy egret (*Egretta thula*). Even today, many serve as symbols of cultural identity, conservation organisations and environmental programmes (Bibby 2002). Beyond their cultural significance, it is widely considered that the numbers of waterbirds using a site is an indicator of quality/biological importance (Gilissen et al. 2002, Kushlan et al. 2002). Thus, their conservation and management has implications for wetland biodiversity in general, not least with regards to the threats faced by wetlands themselves.

Third, under international law, the UK is bound to a number of conservation Directives and Conventions. Thus, there is a legal requirement in the UK to provide for the protection and management of waterbird populations and their associated wetland habitats. In particular, the combined requirements of the Ramsar Convention on Wetlands of International Importance and the EU Wild Birds Directive (79/409/EEC), mean that the UK is under obligation to provide for the long-term protection and persistence of these species, and for the integrity and quality of the wetland habitats upon which they depend.

The Ramsar Convention requires that signatories identify at least one wetland area as internationally important (Ramsar Site). Similarly, all EU countries are legally obliged to identify nationally important sites as Special Protection Areas (SPAs) (discussed in detail in Chapter 2). In recognition of the international importance of wetlands in the UK for their migratory waterbird populations, and of the need for international cooperation successfully to protect these species, the UK also ratified the Agreement on the Conservation of African-Eurasian Migratory Waterbirds (AEWA). As part of the Bonn Convention on the Conservation of Migratory Species of Wild Animals, the AEWA represents a concerted effort on behalf of 28 Eurasian and 17 African nations to ensure the conservation and management of these species. The agreement, which came into force in 1999, covers 235 species of birds ecologically dependent on wetlands and requires that signatories make suitable efforts to identify sites and habitats for migratory waterbirds. Further, the award of a Global Environment Facility (GEF) grant of approximately US$350,000 recently endorsed its development. The full project, coordinated by Wetlands International, in close cooperation with the Ramsar Bureau, AEWA secretariat and BirdLife International, will support countries in priority capacity building activities to assist the joint implementation of their obligations under the Ramsar Convention and the AEWA.

1.4 Thesis aims and objectives

Explicit in the Ramsar Convention, the AEWA and the EU Birds Directive is that signatories/member countries must take the requisite measures to monitor the performance of those sites identified as priorities for conservation. In this respect, the Nature Conservancy Council (NCC) published the first review of the SPA network in 1990 (Stroud et al. 1990). Given the lack of data for many species, the government deemed the report insufficient and requested a further review to be carried out by the Joint Nature Conservation Committee (JNCC). In addition, the government requested that the JNCC provide definitive guidelines for the selection of additional sites. These guidelines were eventually published in 1999 (JNCC 1999), followed by a broad overview of all terrestrial SPAs in 2001, based on available data for the five-year period, 1990/91-1994/95 (Stroud et al. 2001).

This thesis, at the most basic level, provides a detailed and more contemporary view of the performance of sites classified as SPAs classified for their wintering waterbirds (Chapters 2 and 3). Further, given that many SPAs are also recognised under the Ramsar Convention as wetlands of international importance and are inevitably of national significance (see Chapter 2), the performance of those sites designated as Ramsar Sites is evaluated together with that of the wetland SPAs.

Further to the in-depth analysis of the performance of the SPA/Ramsar Site network in Great Britain, this thesis also explores alternative methods for the selection of priority sites for waterbirds, to enable the efficient and effective protection of all target species listed under the EU Birds Directive and the Ramsar Convention (Chapters 4-7). In this respect, numerous authors have used data sets of varying quality to explore and test theories and predictions relating to the effective and efficient selection of hypothetical networks of protected areas, yet relatively few have attempted to apply these same approaches to real-world problems (discussed in detail in Chapter 10). Furthermore, despite repeated and increasingly urgent calls for the evaluation and critical assessment of existing protected areas, evaluations of this type remain rather rare (although see Table 1.4).

Table 1.4: Examples of published studies of the performance of existing networks of protected areas, the features analysed, geographic region considered, and main conclusions.

Reference	Geographic region	Feature	Main conclusions
Pressey and Nicholls (1989a)	New South Wales, Australia	Land Systems	Inclusion of the existing reserves decreases the efficiency of the minimum set (5.7% of the study area increased to 8.3%).
Castro (1992)	Iberian Peninsula	Plants	97 additional squares needed to represent all species at least once.
Nilsson and Götmark (1992)	Sweden	Landscape and habitat types	<1% of the farmland and <2% of the coniferous forests were represented in existing protected areas. At least half the species for these areas are not represented in existing protected areas.
Rebelo and Siegfried (1992)	Cape Region, South Africa	Plants	Existing reserves contain no more species than predicted by a null model.
Sætersdal et al. (1993)	Western Norway	Plants, birds	The 12 sites in the existing reserve system are, to a large extent, duplicating each other, and many species are missed from the total species list.
Fearnside and Ferraz (1995)	Brazil	Vegetation	Only ⅓ of terrestrial vegetation zones are protected. Need to extend the reserve network by 67%.
Lombard (1995)	South Africa	Snakes	Between 63 and 78% of a near-minimum set are reserves.
Lombard et al. (1995)	South Africa	Six vertebrate taxa (104 species)	85% of the hotspots of endemism, richness and rarity coincide with existing reserves. >80% of 5 taxa are represented within the existing network and only eleven species may currently be unprotected.
Pressey et al. (1996)	New South Wales, Australia	Land Systems	Need to expand the existing network by 78% to include all systems at least once. Existing reserves lower the efficiency with which all land systems can be represented at least once (31% more sites, 44% larger area).

Williams et al. (1996)	Great Britain	Birds	20 additional sites are needed to represent all species at least once.
Howard et al. (1997)	Uganda	Woody plants, birds, small mammals, butterflies and large moths	>7% of indicator species represented in existing reserves. Adding 11 forests would increase the representation to 95%.
Khan et al. (1997)	Meghalaya State, NE India	Plants	Reserves are insufficient to protect diversity.
Barnard et al. (1998)	Namibia	Desert vegetation types	The existing network covers only 13.8% of the land area. Four of the 14 desert vegetation types are comprehensively protected (67-94%), yet six have <5% of their extent within protected areas.
Freitag et al. (1998)	Transvaal Region, South Africa	Mammals	Need to expand the existing network by adding 9 cells (25% more sites).
Jaffre et al. (1998)	New Caledonia	Plants	83% do not occur within protected areas.
Nantel et al. (1998)	Newfoundland, Canada	Plants	43% of species are not found within the protected areas.
Araújo (1999)	Portugal	Vertebrates, Plants	The existing protected areas were significantly more efficient than random sets, but significantly less efficient than complementary sets.
Rodrigues et al. (1999)	Scotland	Wetland plants	The current SSSI network has been selected in an inefficient manner. Although, it performed considerably better than a random set of sites where at least two representations of each species were required.
Hopkinson et al. (2000a)	Great Britain	Ten taxonomic groups	National Nature Reserves and the combined network of National Nature Reserves, Royal Society for the Protection of Birds (RSPB) sites and Sites of Special Scientific Interest (SSSIs) showed significantly greater overlap with hotspots compared with random sets of sites. 79% of sites are present in only one of the networks. >94%

			of species in each taxonomic group were in the combined network.
Powell et al. (2000)	Costa Rica	Life zones as indicators of terrestrial biodiversity	The addition of approximately 400,000ha to the existing park system by expanding 12 existing protected areas and adding one new one would extend the numbers of life zone protected from nine to 19.
(Andelman and Willig 2002)	Paraguay	Bats	The existing network had relatively low value for the conservation of the nation's bat fauna, particularly the rare species.
Sierra et al. (2002)	Continental Ecuador	Terrestrial ecosystems	The current network is highly inefficient and only 12 of the 22 ecosystems are represented at ≥ 50% of their area. Seven ecosystems are missing from the network altogether.
Rouget et al. (2003)	Cape Floristic Region, South Africa	Vascular plants	Approximately 20% of the region is under some form of protection. The representation bias towards upland areas has seriously constrained the representation of biodiversity pattern and process.

This thesis will, therefore, address three additional issues:

1. The effectiveness and utility of alternative selection methods, and in particular the use of linear programming for real-world conservation issues (Chapters 4-6);
2. The distribution patterns and population dynamics of selected species, and their implications for the selection and management of SPAs and Ramsar Sites, both in Great Britain and across the EU (Chapters 7-9);
3. Suggestions for the improvement of the current network of protected areas in the UK and across the EU and to provide recommendations and suggestions for waterbird conservation in general.

1.5 Data

Unless otherwise stated, the analyses included within Chapters 2-9 were conducted using data from the Wetland Bird Survey (WeBS). This scheme, a joint venture by the British Trust for Ornithology (BTO), Wildfowl and Wetlands Trust (WWT), Royal Society for the Protection of Birds (RSPB), and the Joint Nature Conservation Committee (JNCC), aims to monitor the populations of non-breeding waterbirds in the UK (Kirby 1995, Cranswick et al. 1997, Pollitt et al. 2003). Specifically, the survey relies on volunteers visiting sites on pre-selected dates on one occasion each month, with emphasis on winter months September to March, and recording the numbers of all waterbird species present. These individual counts are generally made in the morning and normally at high tide on estuarine sites. More than 2000 wetlands are included in the scheme annually, and over 8000 have been counted at least once since 1960 (Cranswick et al. 1997; Fig. 1.1). Each WeBS recording unit from which the monthly counts for each species individually are taken forms part of a larger WeBS site (representing a distinct wetland area). These consolidated sites (henceforth simply 'sites') in the majority of cases correspond closely to actual wetland site/PS boundaries, therefore, annual site counts referred to throughout this thesis reflect numbers for these larger sites and not the individual recording units.

For all analyses, unless otherwise stated in the Methods section of the relevant chapter, the identity and numbers of WeBS sites ($N = 1962$) and protected sites (SPAs/Ramsar Sites; $N = 138$) remained constant in all years for each species analysed. These WeBS sites are those for which at least 60% of the monthly and annual abundance records were available. Missing values were imputed using linear interpolation (SPSS version 11), based on the method adopted by WeBS for the calculation of annual population indices for Great Britain (Kershaw and Cranswick 2003, Pollitt et al. 2003). Finally, given that WeBS counts in Northern Ireland did not begin until 1985 all analyses use data from Great Britain only (excluding the Channel Islands and the Isle of Man).

Figure 1.1: Map showing the location of all Wetland Bird Survey Sites (navy blue squares) and those classified as SPAs/Ramsar Sites (red squares). NB. Only those sites included in subsequent chapters are shown.

Given the constraints imposed by variation in the numbers of sites classified as nationally important (see Chapter 2), the occurrence of missing counts and the year in which counts began by WeBS for each waterbird species (i.e. see Chapter 2), the data period used and the numbers and identity of the individual species analysed for each chapter were chosen so as to maximise the power of the statistical tests applied, and as a result are not consistent throughout this thesis (Table 1.5).

Although WeBS is considered one of the most comprehensive datasets currently available, some members of the scientific community remain sceptical about its reliability, particularly given the involvement of volunteers. Indeed, some variation in the ability of amateur bird watchers to detect, identify and record individual species and their abundance is inevitable (Foster-Smith and Evans 2003). Nonetheless, it has been shown that volunteers, given the appropriate amount of information, are able to identify required species to a level equivalent to experts. For example, comparison of the ability of experts and volunteers to correctly record and identify common littoral organisms of the Isle of Cumbrae, Scotland showed that, although mistakes were made, these were not significantly different to those made by the experts (Foster-Smith and Evans 2003).

Table 1.5: The data period and number of species included in the analyses presented in each data chapter (2-9) of this thesis. NB. The identity of each species is presented in the Methods section of the relevant chapter.

Chapter	Data period	Number of species
Chapter 2	1960/61-1998/99	21
Chapter 3.1	1975/76-1998/99	21
Chapter 3.2	1990/91-1998/99	13
Chapter 4	1976/77-1998/99	17
Chapter 5	1981/82-1998/99	17
Chapter 6	1989/90-1998/99	17
Chapter 7	1985/86-1998/99	14
Chapter 8	1980/81-1998/99	19
Chapter 9	1993/94-1998/99	21

Aside from sources of error introduced through the use of volunteers, there are a number of systematic biases to WeBS that should be borne in mind (described in detail in Kershaw and Kranswick 2003, Pollitt et al. 2003). These relate, in particular, to the incomplete WeBS coverage of all wetland areas (missing counts and that not all wetland sites are covered) and the distribution patterns of individual species. First, the coverage of widely dispersed species (e.g. little grebe, cormorant, mute swan, mallard, teal and goosander) is likely to be under-represented given the concentration of efforts towards estuarine habitats and large, standing waters. Second, numbers of cryptic or secretive species (e.g. little grebe and teal) are likely to be overlooked given the problems associated with their detection. Third, numbers of passage species are also likely to be under-estimated given the high turnover of individuals in a short period of time. By contrast, counts of the numbers of particularly mobile flocks are likely to be over-estimates, particularly where individuals move between sites, which may result in the same individuals being counted more than once for a given month.

Nonetheless, aside from these problems, WeBS is a remarkable and almost unparalleled resource. In comparison with other datasets, WeBS coverage (both temporally and spatially) of species and sites is exceptional. Furthermore, all records received from volunteers are rigorously checked to identify potential errors, thereby improving overall dataset accuracy.

1.6 Thesis Outline

The remainder of this thesis is structured into nine chapters:

Chapter 2 explores the current situation as regards the network of Special Protection Areas (SPAs) and Ramsar Sites in Great Britain for selected, regularly occurring waterbird species.

Chapter 3 is split into two main sections, the first of which investigates the performance of this network of protected areas with respect to random sets of sites and annual population fluctuations. The second section deals with comparisons between protected and non-protected (as SPAs/Ramsar Sites) areas.

Chapter 4 addresses the effectiveness of the SPA/Ramsar Site network in comparison with alternative priority-site selection strategies.

Chapter 5 explores the implications of changes to the characteristics of data included in a site selection procedure for waterbird species on the performance of hypothetical reserve networks.

Chapter 6 is concerned with the exploration of network flexibility and irreplaceability using linear programming algorithms.

Chapter 7 deals with identifying possible habitat management conflicts between waterbird species and the implications of the inclusions of these conflicts on the results of priority site selection.

Chapter 8 explores the buffer effect concept and its implications for the selection and management of SPAs and Ramsar Sites.

Chapter 9 looks at the capacity of protected areas to buffer annual variability in the numbers of individuals with respect to EU level and also national-level conservation policy requirements.

Chapter 10 looks at the probable reasons for the gap between theoretical and practical conservation biology, and provides suggestions as to how to overcome these difficulties. Finally, this thesis ends with a discussion of the role of conservation biology and conservation biologists now and into the future.

CHAPTER 2: CURRENT NETWORK STATUS

"Acquiring a nature reserve looks like an achievement, but it will come to naught unless the place is looked after properly (Marren, 2002)."

Various stakeholders have a vested interest in the effective and efficient performance of protected areas for conservation. Indeed, investors (e.g. government departments and conservation agencies), sponsors (e.g. Northumberland Water, Tesco and Shanks McEwen sponsor biodiversity action plans (BAPs) for the roseate tern (*Sterna dougallii*), skylark (*Alauda arvensis*) and corncrake (*Crex crex*) respectively) and local communities, habitually provide funding, manual labour and technical advice. Each of these groups requires/expects that protected areas are managed effectively to ensure good value for money and a return for their investment. It is, therefore, essential that protected area networks be regularly evaluated to identify and respond to problems and inadequacies before the condition worsens beyond the capabilities of managers to restore or improve the situation. In this respect, Chapters 2 and 3 look in detail at the status of the current (1998/99) network of SPA and Ramsar Sites in Great Britain. Specifically, Chapter 2 provides an overview of the numbers and representation of a selection of wintering waterbird species. Chapter 3 then evaluates the performance of this network over time with respect to national trends and random sets of sites (Chapter 3.1). Chapter 3.2 compares the rates of population size change and mean numbers of individuals over time on protected and non-protected wetlands.

Size matters:

The value of small populations for wintering waterbirds.

2.1 Introduction

The designation of Ramsar sites under the Ramsar Convention on Wetlands of International Importance and the classification of Special Protection Areas (SPAs) under the Directive on the Conservation of Wild Birds (the 'Birds Directive'; 79/409/EEC) are important vehicles for the conservation of sites important for waterbird conservation in the UK. Indeed, the aggregate distribution of many waterbirds during the winter (non-breeding period) means that site designation and management acts as an effective conservation mechanism for many such species (Kershaw and Cranswick 2003). Adopted in Iran in 1971, the Ramsar Convention was the first of the modern global intergovernmental treaties on the conservation and wise use of natural resources. Through the designation, management and monitoring of internationally important wetlands across the world, the aim is to achieve: '...the conservation and wise use of all wetlands through local, regional and national actions and international cooperation, as a contribution towards achieving sustainable development throughout the world' (Ramsar Convention Bureau 2002a).

Contracting Parties to the Ramsar Convention are expected to promote wetland conservation within their borders through implementation of the 'wise-use principle' outlined in the Convention. Coupled with this, continued dialogue between Contracting Parties and regular meetings further encourages countries seeking to improve on desired targets. Fundamentally, such discourse aims to promote awareness of the intrinsic value of wetlands and thus the urgency for a concerted effort towards their protection and management, particularly where conservation priorities transcend political borders. One of the basic commitments undertaken by Contracting Parties is the designation of at least one wetland for the List of Wetlands of International Importance. Since coming into force in 1975, 1,252 wetlands, with a collective area exceeding 107.5 million ha, have been officially designated as Ramsar Sites by 136 contracting parties across six administrative regions (Africa, Asia, Europe, north America, Oceania and the Neotropics). Of these designations, 59.9% are located within Europe, 11.7% in Asia, 9% in Africa, 8.7% in the Neotropics, 5.7% in Oceania and 5% in North America (Ramsar Convention Bureau 2002b).

Ramsar sites are identified on the basis of internationally agreed criteria, adopted by the 4th, 6th and 7th Conferences for Contracting Parties (Ramsar Convention Bureau 1990, 1996, 1999). Several of these relate specifically to waterbirds, such that sites that regularly hold a total of 20,000 or more waterbirds (Criterion 5 as defined by the Conference of Parties to the Ramsar Convention; Ramsar Convention Bureau 1980) or that regularly hold at least 1% of the individuals in a biogeographic population of one species or subspecies of waterbird, qualify as Ramsar Sites (Criterion 6; Ramsar Convention Bureau 1999). Globally, 694 sites have been designated explicitly using selection criteria 5 or 6.

Table 2.1: Date of entry to the convention, numbers of Ramsar Sites and total area (ha) of all designated sites for the 136 contracting parties to the Ramsar Convention on Wetlands of International Importance. [a]No information. (Source http://www.ramsar.org/key_cp_e.htm).

Country	Date of entry	Ramsar sites	Surface area
Albania	29.02.96	1	20,000
Algeria	04.03.84	13	1,866,340
Argentina	04.09.92	12	3,074,589
Armenia	06.11.93	2	492,239
Australia	21.12.75	63	7,287,645
Austria	16.04.83	11	117,952
Azerbaijan	21.05.01	2	99,560
Bahamas	07.06.97	1	32,600
Bahrain	27.02.98	2	6,810
Bangladesh	21.09.92	2	605,500
Belarus	25.08.91	7	276,307
Belgium	04.07.86	6	7,935
Belize	22.08.98	1	6,637
Benin	24.05.00	2	139,100
Bolivia	27.10.90	8	6,518,073
Bosnia and Herzegovina	01.03.92	1	7,411
Botswana	09.04.97	1	6,864,000
Brazil	24.09.93	7	6,346,215
Bulgaria	24.01.76	10	20,306
Burkina Faso	27.10.90	3	299,200
Burundi	05.10.02	1	1,000
Cambodia	23.10.99	3	54,600
Canada	15.05.81	36	13,051,501
Chad	13.10.90	2	1,843,168
Chile	27.11.81	7	100,174
China	31.07.92	21	2,547,763
Colombia	18.10.98	2	439,000
Comoros	09.06.95	1	30
Congo	18.10.98	1	438,960
Costa Rica	27.04.92	10	317,530
Côte d'Ivoire	27.06.96	1	19,400
Croatia	25.06.91	4	80,455
Cuba	12.08.01	6	1,188,411
Cyprus	11.11.01	1	1,585
Czech Republic	01.01.93	10	41,861
Democratic Republic of Congo	18.05.96	2	866,000
Denmark	02.01.78	38	2,078,823
Djibouti	22.03.03	1	3,000
Dominican Republic	15.09.02	1	20,000

Ecuador	07.01.91	10	113,634
Egypt	09.09.88	2	105,700
El Salvador	22.05.99	1	1,571
Estonia	29.07.94	10	215,950
Finland	21.12.75	11	138,746
France	01.12.86	18	795,085
Gabon	30.04.87	3	1,080,000
Gambia	16.01.97	1	20,000
Georgia	07.06.97	2	34,223
Germany	26.06.76	31	828,931
Ghana	22.06.88	6	178,410
Greece	21.12.75	10	163,501
Guatemala	26.10.90	4	502,707
Guinea	18.03.93	12	4,779,061
Guinea-Bissau	14.05.90	1	39,098
Honduras	23.10.93	5	179,680
Hungary	11.08.79	21	154,147
Iceland	02.04.78	3	58,970
India	01.02.82	19	648,507
Indonesia	08.08.92	2	242,700
Iran, Islamic Republic of	21.12.75	21	1,475,720
Ireland	15.03.85	45	66,994
Israel	12.03.97	2	366
Italy	14.04.77	46	57,137
Jamaica	07.02.98	1	5,700
Japan	17.10.80	13	84,089
Jordan	10.05.77	1	7,372
Kenya	05.10.90	4	90,969
Kyrgyz Republic	12.03.03	1	[a]
Latvia	25.11.95	3	43,300
Lebanon	16.08.99	4	1075
Libyan Arab Jamahiriya	05.08.00	2	[a]
Liechtenstein	06.12.91	1	101
Lithuania	20.12.93	5	50,451
Luxembourg	15.08.98	1	313
Madagascar	25.01.99	2	53,095
Malawi	14.03.97	1	224,800
Malaysia	10.03.95	1	38,446
Mali	25.09.87	3	162,000
Malta	30.01.89	2	16
Mauritania	22.02.83	3	1,231,100
Mauritius	30.09.01	1	26
Mexico	04.11.86	7	1,103,976
Monaco	20.12.97	1	10
Mongolia	08.04.98	6	630,580
Morocco	20.10.80	4	14,350
Namibia	23.12.95	4	629,600
Nepal	17.04.88	1	17,500
Netherlands	23.09.80	38	691,228
New Zealand	13.12.76	5	38,868
Nicaragua	30.11.97	8	405,691
Niger	30.08.87	4	715,302
Nigeria	02.02.01	1	58,100

Norway	21.12.75	37	116,369
Pakistan	23.11.76	16	283,952
Palau	18.02.03	1	493
Panama	26.11.90	3	110,984
Papua New Guinea	16.07.93	2	594,924
Paraguay	07.10.95	4	775,000
Peru	30.03.92	8	6,759,388
Philippines	08.11.94	4	68,404
Poland	22.03.78	8	90,455
Portugal	24.03.81	12	66,096
Republic of Korea	28.07.97	2	960
Republic of Moldova	20.10.00	1	19,152
Romania	21.09.91	2	664,586
Russian Federation	11.02.77	35	10,323,767
Saint Lucia	19.06.02	2	85
Senegal	11.11.77	4	99,720
Sierra Leone	13.04.00	1	295,000
Slovak Republic	01.01.93	12	38,208
Slovenia	25.06.91	2	955
South Africa	21.12.75	17	498,721
Spain	04.09.82	38	158,143
Sri Lanka	15.10.90	2	7,607
Suriname	22.11.85	1	12,000
Sweden	21.12.75	51	514,500
Switzerland	16.05.76	8	7,946
Syrian Arab Republic	05.07.98	1	10,000
Tajikistan	18.11.01	5	94,600
Thailand	13.09.98	10	370,600
The FYR of Macedonia	08.09.91	1	18,920
Togo	04.11.95	2	194,400
Trinidad and Tobago	21.04.93	1	6,234
Tunisia	24.03.81	1	12,600
Turkey	13.11.94	9	159,300
Uganda	04.07.88	1	15,000
Ukraine	01.12.91	22	716,250
United Kingdom	05.05.76	169	859,023
United Republic of Tanzania	13.08.00	3	4,271,516
United States of America	18.04.87	19	1,192,730
Uruguay	22.09.84	1	407,408
Uzbekistan	08.02.02	1	31,300
Venezuela	23.11.88	5	263,636
Viet Nam	20.01.89	1	12,000
Yugoslavia	28.07.77	4	39,861
Zambia	28.12.91	2	333,000
Former USSR	[a]	5	1,559,500

In contrast to the detailed criteria set out by the Ramsar Convention Bureau for the designation of Ramsar sites, the European Union Birds Directive does not state specifically how SPAs are to be selected, rather the exact mechanisms depend on the Member States (Stroud et al. 2001). Indeed, the Directive states simply that: 'Member States shall classify in particular the most suitable territories in number and size as special protection areas for the conservation of these species, taking into account their protection requirements in the geographical sea and land area where this Directive applies'.

In the UK, sites qualify for SPA status under guidelines published by the Joint Nature Conservation Committee (JNCC) in 1999. These follow a two-stage process and include a number of possible grounds for classification, namely:

- Stage 1.1: An area used regularly by 1% of the Great Britain (or all-Ireland) population of any species listed as rare or vulnerable in Annex I of the Birds Directive (Article 4.1) in any season (i.e. whooper swan and Bewick's swan).
- Stage 1.2: An area used regularly by greater than 1% of the biogeographical population of those species listed as regularly occurring migratory species (Article 4.2) in any season (i.e. goldeneye and tufted duck).
- Stage 1.3: An area used regularly by more than 20,000 waterfowl (waterfowl as defined by the Ramsar Convention) in any season.
- Stage 1.4: To provide an adequate suite of sites for an Annex I or regularly occurring migratory species where the application of Stage 1.1, 1.2 and 1.3 guidelines for a species does not yield an adequate suite of sites for the conservation of that species (to target, for example, wider-ranging or thinly dispersed species).

Following the application of Stage 1 guidelines, ecological features with reference to target species are evaluated/considered for each proposed site to further refine the selection, namely: population size and density, species range, breeding success, history of occupancy, multi-species areas, naturalness and severe weather refuges (JNCC 1999). Those sites considered "most suitable" should eventually be classified as SPAs. Finally (on those sites selected as assemblages under guideline 1.3), those species that regularly occur in nationally important numbers within sites (1% or more of the national population size present, or 50 birds where the national 1% threshold is less than 50 individuals), or which number 2000 or more individuals within an assemblage, are listed as the main component species of the assemblage. This Listed Site (LS) suite, where each component site should be managed so as to maintain numbers of that species, should fulfil the relevant site-protection requirements for that particular species or population (Stroud et al. 2001). The LS for each species, therefore, consists of those sites that have been selected as "most favourable" for the species concerned in the context of the Birds Directive. Under national legislation, those sites supporting nationally important numbers of birds qualify as Sites of Special Scientific Interest (SSSIs).

As of September 2001 there were 140 Ramsar sites in the UK, the first of which was designated in 1976, and 243 SPAs (both wetland and non-wetland) (JNCC 2001), many having been designated very recently, with 96 of the SPAs also being designated as Ramsar sites. The first SPA was classified in 1982 and the first wetland SPA in 1985 (JNCC 2001). The distribution of SPAs is uneven across the UK, with 136 sites in Scotland, 80 in England, 18 in

Wales and 12 in Northern Ireland (these totals add up to more than 243 due to cross-border sites). Over the winter period, the SPA network alone supports in excess of 2 million non-breeding waterbirds, approximately 40% of all waterbirds present in the UK (Stroud 2002).

An attractive feature of the numerical SPA/Ramsar Site selection guidelines is their simplicity, which enables them to be applied widely. However, they may not sufficiently take account of many features of wetland sites that potentially contribute to their importance for the long-term maintenance of waterbird populations, nor do they necessarily account for the seasonal and long-term dynamics of species numbers (Kershaw and Cranswick 2003, Rehfisch et al. 2003). Moreover, the emphasis is placed on a site-by-site approach rather than one that views the conservation network in its entirety. The focus tends towards single species numbers, and as a result incorporation of maximum biodiversity (although not a specific requirement of the Birds Directive) is incidental rather than an explicit selection objective (Lee et al. 2001).

In this Chapter, I examine the numbers of individuals of selected waterbird species that are found in the overall wetland SPA/Ramsar Site network (henceforth referred to as the Protected Sites (PS) network) in Great Britain (as of 1998/99) and on the suites of SPAs that have been classified for individual species (LS). The current effectiveness of these protected area networks is evaluated in relation to the patterns of distribution shown by different species, with the emphasis on the numbers of birds of each of the species contained within the national network rather than looking at explicit selection criteria detailed by the Birds Directive, or the extent to which these have been successfully implemented.

2.2 Methods

Data

Twenty-one species of waterbird considered well represented by WeBS and for which this is the principal source of data used for the selection of SPA and Ramsar sites in the UK, were analysed separately (Table 2.2). Data for the years 1960/61 (earliest computerised waterbird data) to 1998/99 (latest available data in September 2000) inclusive were included for little grebe (data only available from 1985/86), great crested grebe (data only available from 1982/83), cormorant (data only available from 1986/87), whooper swan, Bewick's swan, European white-fronted goose, dark-bellied brent goose, shelduck, wigeon, gadwall, teal, mallard, pintail, shoveler, pochard, tufted duck, goldeneye, smew, red-breasted merganser, goosander and coot (data only available from 1982/83).

To qualify as an SPA under Stage 1.3 of the JNCC site-selection guidelines requires a site to support ≥20,000 waterbirds irrespective of the actual species present (i.e. the total across all WeBS species present (excluding waders) and not simply those listed as targets by the Ramsar Convention and/or the Birds Directive; see Pollitt et al. 2003 for details).

Table 2.2: For each species of waterbird included in this thesis: the most recently available population trend across the western Palaearctic (Gilissen et al. 2002), expressed as STA (Stable), INC (Increasing), DEC (decreasing) or ? (Uncertain); Species of European Conservation Concern (SPEC), expressed as SPEC 3 (species not concentrated in Europe but with an unfavourable conservation status) or SPEC 4 (species concentrated in Europe with a favourable conservation status) (JNCC 2002); European Threat Status, expressed as S (Secure), L (Localised) and V (Vulnerable) (**http://www.wetlands.org/pubs&WPE.htm**); Species included in the Ramsar Convention on Wetlands of International Importance in Annex I or Annex II (**http://www.ramsar.org**); Species included in the Bern Convention under Appendix II or III (**http://conventions.coe.int**); Species included in the Bonn Convention under Appendix II (species which would benefit from international cooperation in their conservation and management) (**http://www.wcmc.org.uk/cms**); and species included in the Agreement on the Conservation of African-Eurasian Migratory Waterbirds (AEWA) (**http://www.unep-aewa.org**).

Species		Trend	SPEC	European threat status	Ramsar Convention	Bern Convention	Bonn Convention	AEWA
Little grebe	*Tachybaptus ruficollis*	STA		S	II	II		
Great crested grebe	*Podiceps cristatus*	INC		S	II	III		
Cormorant	*Phalacrocorax carbo*	INC		S	II	II		
Mute swan	*Cygnus olor*	INC		S	II	III	II	*
Bewick's swan	*Cygnus columbianus bewickii*	DEC	3	L	I	II	II	*
Whooper swan	*Cygnus cygnus*	INC	4	S	I	II	II	*
European white fronted goose	*Anser albifrons albifrons*	INC		S	II	III	II	*
Canada goose	*Branta canadensis*							
Dark-belied brent goose	*Branta bernicla bernicla*	DEC	3	V	II	III	II	*
Shelduck	*Tadorna tadorna*	STA		S	II	II	II	*
Wigeon	*Anas penelope*	INC?		S	II	III	II	*
Gadwall	*Anas strepera*	INC	3	V	II	III	II	*
Teal	*Anas crecca*	STA		S	II	III	II	*
Mallard	*Anas platyrhynchos*	DEC		S	II	III	II	*

Pintail	*Anas acuta*	DEC		S	II	III	II	*
Shoveler	*Anas clypeata*	STA		S	II	III	II	*
Pochard	*Aythya ferina*	STA	4	S	II	III	II	*
Tufted duck	*Aythya fuligula*	INC		S	II	III	II	*
Goldeneye	*Bucephala clangula*	INC		S	II	III	II	*
Smew	*Mergellus albellus*	INC	3	V	II	II	II	*
Red-breasted merganser	*Mergus serrator*	INC		S	II	III	II	*
Goosander	*Mergus merganser*	INC		S	II	III	II	*
Coot	*Fulica atra*	INC		S	II	III		*

Therefore, for each wetland site, in addition to the 21 species separately, totals were taken across all WeBS species.

National totals

For each of the 21 species and for all WeBS species together, following the procedure adopted by WeBS (Pollitt et al. 2000), the national total (total number of individuals in Great Britain) for a given year and month was take to be the sum of the individual monthly counts across all WeBS sites in Great Britain. The peak monthly count in each year was then taken to represent the national total for each species. Where national counts are referred to in the following text these relate to WeBS totals for each species and may be less than other published national totals given the limitation of WeBS coverage (see Chapter 1 for discussion).

Individual site totals

Individual site counts were taken as the maximum number of individuals recorded on each site between September and March for each year 1960/61-1998/99. Annual PS totals (all sites classified as SPAs or Ramsar Sites up to and including those classified in 1998/99), taken as the sum of the individual site counts, were calculated for each count year from 1960/61 to 1998/99. Finally, the peak number of birds on the LS suite for each species in each year (1960/61 to 1998/99) was also calculated from the sum of the individual annual counts for all sites listed as nationally important for a species, and is referred to as the LS total. This will generally be smaller than, although it may equal, the PS total.

Since the numbers of PS have increased over time (Fig. 2.1), the PS and LS totals were calculated for each year using data from only those sites that were formally classified/designated in that year. Table 2.3 lists the number of PS that have been classified in Great Britain for each species as of 1998/99 (JNCC 2001). The PS and LS totals therefore reflect the peak numbers of individuals using sites officially classified/designated as SPAs/Ramsar Sites in Great Britain in each year. It should be borne in mind that the protected site network in Great Britain is constantly evolving and now includes additional sites, but given that data were not available these have not been included here.

Estimates of the current size of the biogeographic populations of each species were obtained from Rose and Scott (1997), the biogeographic population being defined as a species' population inhabiting a defined area or areas that freely interbreeds but tends not to exchange individuals with other populations (Mayr 1970). For example, for white-fronted goose two biogeographic populations are described: Greenland white-fronted goose (*Anser albifrons flavirostris)* and European white-fronted goose (*Anser albifrons albifrons*). For conservation planning it is necessary to consider such biogeographic populations separately given the differing circumstances encountered by each in the wintering and breeding areas (Stroud et al. 1990).

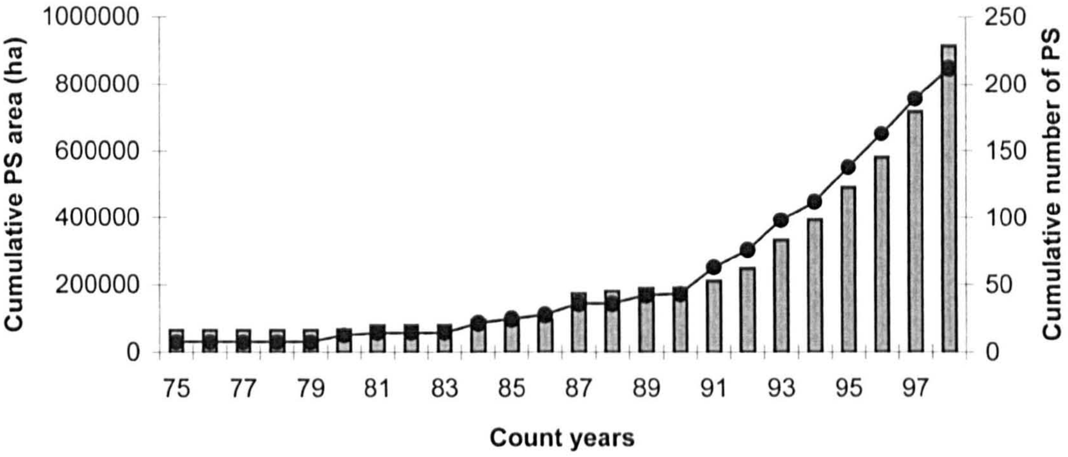

Figure 2.1: The cumulative number of PS (line) classified/designated and their total area (bars) in each count year (1975/76-1998/99).

These biogeographic estimates are calculated from available waterbird count data, drawn primarily from Wetlands Internationals' International Waterbirds Census (which has inputs from WeBS) as well as supplementary data, information and interpretation from Wetlands Internationals' Specialist Groups (Rose and Scott 1997). These population estimates are then used to calculate the 1% thresholds for sites of international importance. International estimates and 1% levels were produced/revised in 1980 (Scott 1980), 1989 (Boyd and Pirot 1989), 1994 (Rose 1994) and 1997 (Rose and Scott 1997), and these were used to derive estimates of the biogeographic population size for the relevant years. However, estimates were unavailable for several species in some or all of the count years used in this investigation; for cormorant, comparisons with the biogeographic population were possible from 1994 and for great crested grebe from 1997, and no biogeographic population estimates are available for little grebe.

Finally, the degree of concordance (Kendall's *w* statistic), using the total annual numbers of all 21 species and annual counts for each species individually, was calculated to determine the degree to which the peaks of abundance occur on the same sites over time. Given the size of the data matrix for each individual species, in each case, 30 random combinations of 500 sites were selected each time.

2.3 Results

Current population distributions

Considering all waterbirds (as defined by the Ramsar Convention, 2002b) included by the WeBS and not just the 21 species examined separately, in the most recent year for which data were analysed (1998/99) the vast majority of sites contained relatively small numbers of birds (Fig. 2.2). Specifically, for 97.5% of the Great Britain WeBS sites less than 20,000 birds were recorded, the threshold level for consideration as a site of international importance as a waterbird assemblage. However, in aggregate, these sites (with <20,000 birds) only contained 32.3% or 1,531,719 of the 4,743,615 waterbirds summed across all sites, the remainder being concentrated on just 2.5% of the sites (Fig. 2.2).

A similar pattern was evident for the 21 selected species of waterbird individually. In 1998/99 most sites contained only small numbers of individuals of a given species. Indeed, for all of these species, >90% of sites individually contained 0-10% of the maximum count for that species (the range of values for individual site counts subdivided into ten equal size classes). For example, this was true for 99.8% of sites for Bewick's swan, 99.1% of sites for dark-bellied brent goose, 98.3% of sites for red-breasted merganser and 97.1% of sites for shoveler. However, for ten species (little grebe, great crested grebe, whooper swan, wigeon, mallard, teal, pochard, tufted duck, goldeneye and coot), those sites that individually contained 0-10% of the maximum count, in aggregate accounted for >50% of the national total, and for four of these species this was >60% (60.1% for great crested grebe, 60.6% for pochard, 64% for tufted duck and 71.5% for goldeneye).

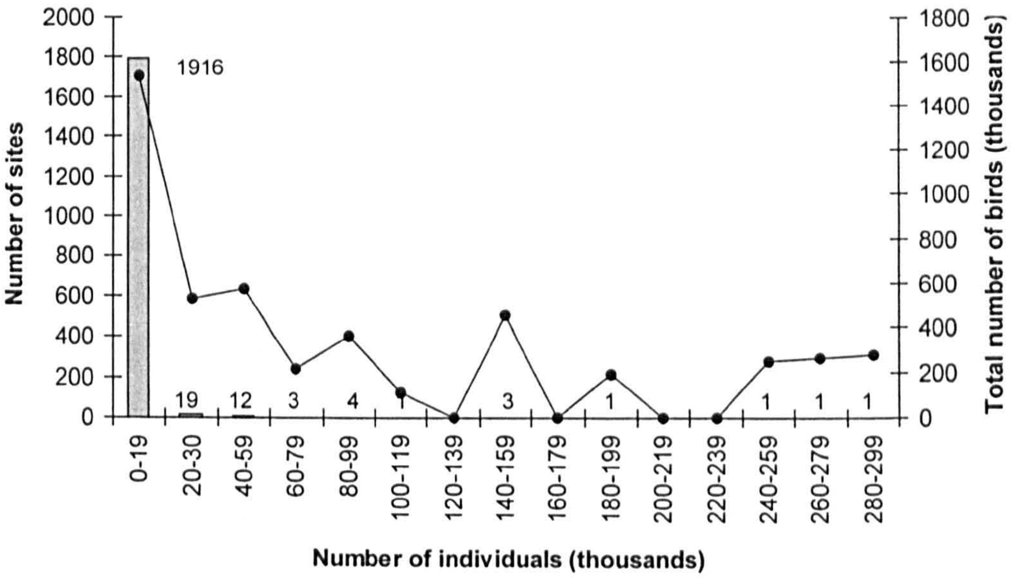

Figure. 2.2: The performance of the PS network in terms of the number of WeBS sites (bars) and the total number of birds present (solid line) (in thousands) for equal sized classes (numbers in thousands) in 1998/99 across all waterbird and wader species included in the WeBS scheme. Numbers indicate the numbers of sites in each category.

Despite in sum containing so many individuals, very few of those sites with low numbers of individuals were protected under either the Birds Directive as SPAs or the Ramsar Convention as Ramsar Sites, demonstrated by the small contribution these sites made to the PS total, compared to the total number of individuals on these sites (Fig. 2.3).

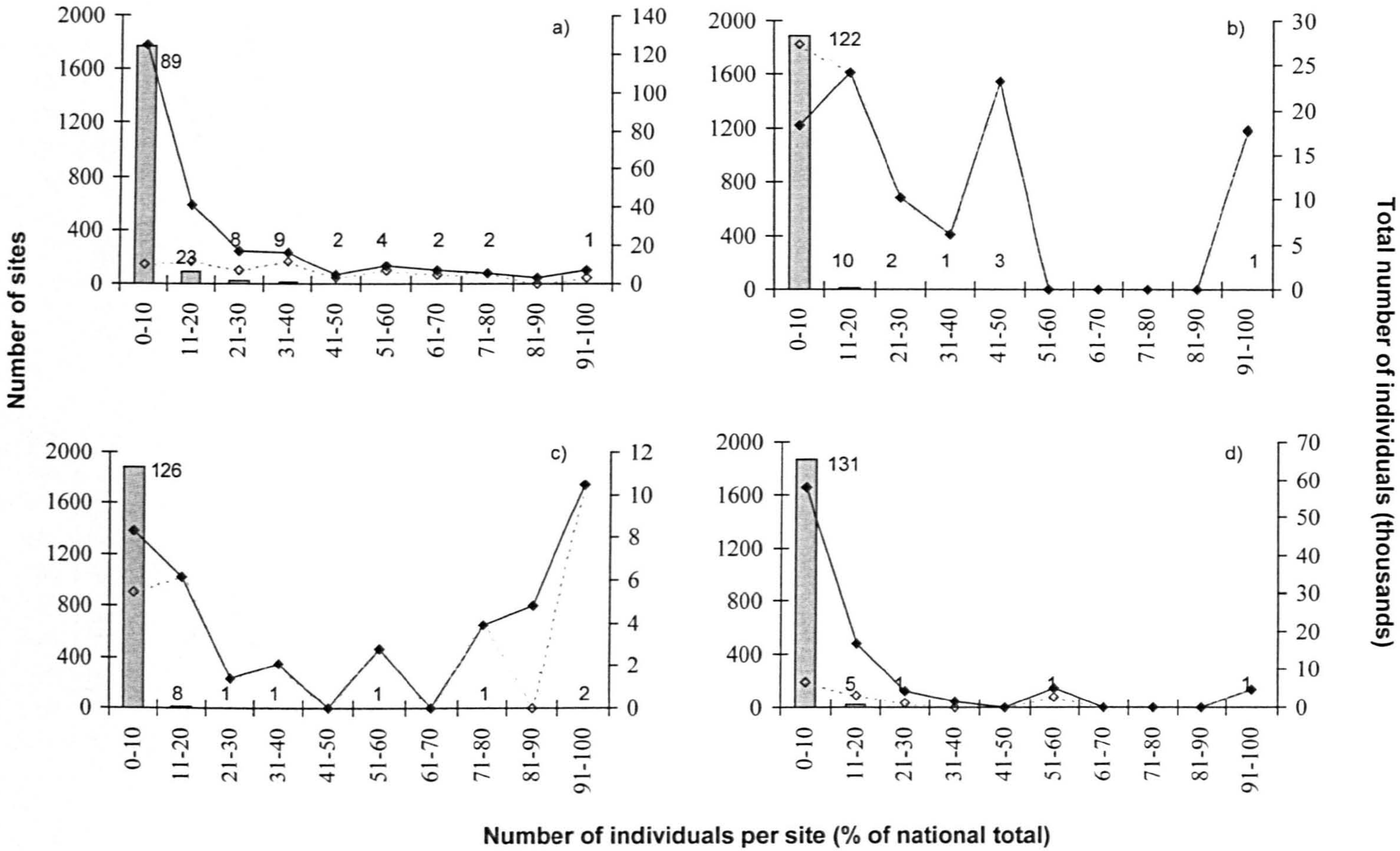

Figure 2.3: The number of WeBS sites for: a) mallard, b) dark-bellied brent goose, c) pintail, and d) tufted duck (bars), the numbers of PS (numbers above bars), the total numbers of birds (solid line), and the PS total (dashed line) in 1998/99 for ten equal sized classes. The categories vary depending on the overall number of individuals recorded for each species, each equate to ten percent of the total range.

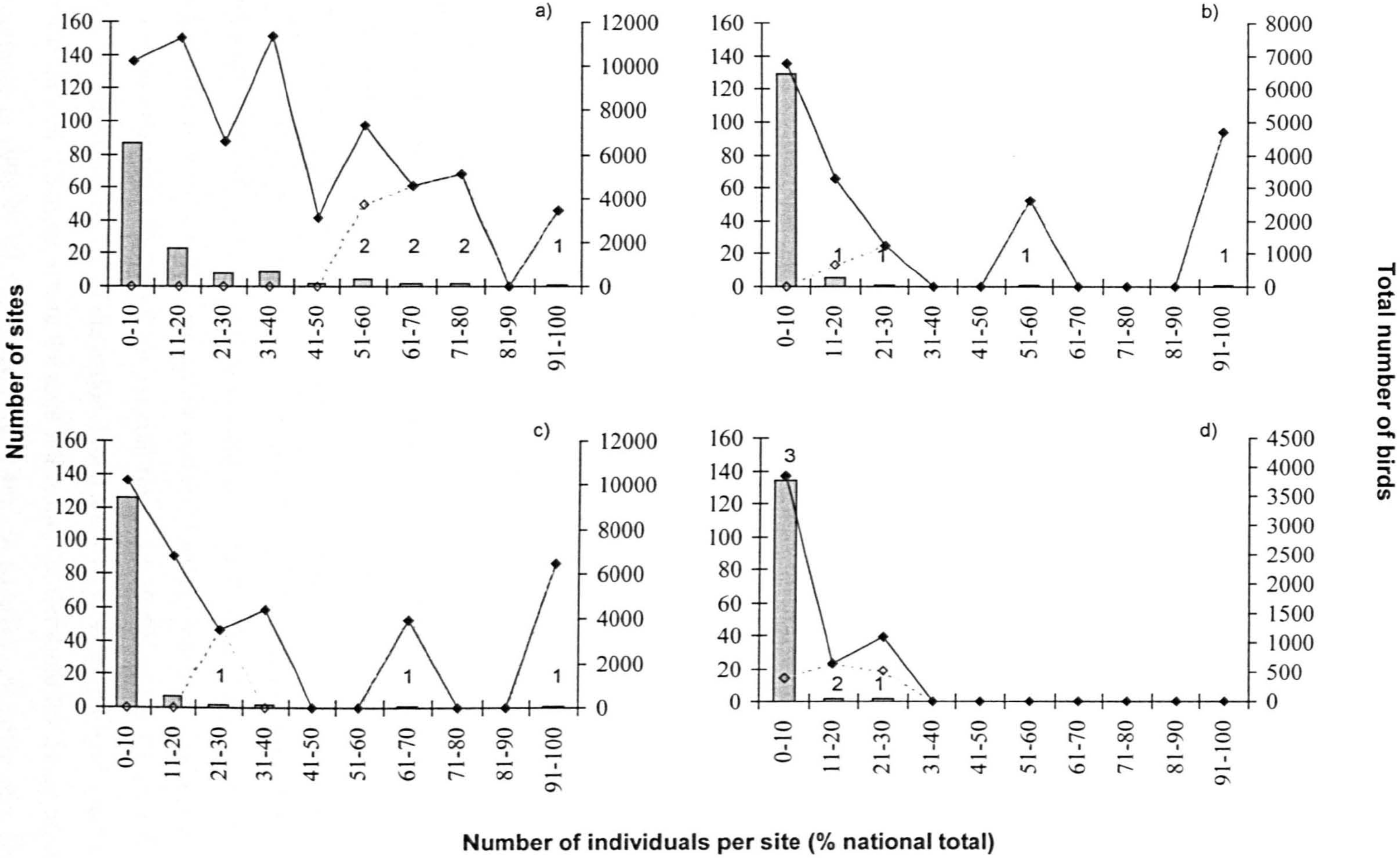

Figure 2.4: The relationship between the number of PS for equal sized classes (bars), the numbers of LS (numbers above bars), the PS total in each category (solid line), and the LS total (dashed line) in 1998/99 for: a) mallard, b) tufted duck, and c) coot. Each percentage category is the same as for Fig. 2.3.

For example, only 4.9% (87, *N* = 1768) of the sites with <= 345 mallard (0-10% range) were included in the PS network. However, these sites in combination account for 53.4% of the national total for mallard. Thus, from a total number of 124,551 mallard on these sites only 10,223 (8.2%) were recorded on protected sites (Fig. 2.3a). In contrast, 70.9% of mallard on sites with between 1,036 and 1,380 birds (31-40% range) were in the PS network, but these sites only hold 6.9% of the national total for this species (Fig. 2.3a). Similarly, sites supporting 0-10% of the maximum count for goldeneye (<233 birds) and pochard (<583 birds) account for 71.5% and 60.5% of the national total respectively and represent 99% of sites for these species; nonetheless, less than 25% of these sites are PS (24% for goldeneye and 14% for pochard). For several species, the distribution of individuals was highly aggregated (Bewick's swan, European white-fronted goose, dark-bellied brent goose and pintail). Additionally, for these species more birds actually occur on the most populous sites than on the sum total of the sites with the lowest numbers of individuals. For example, sites individually with 0-10% of the maximum count, in sum held <20% of the national total for European white-fronted goose (12.8%) and Bewick's swan (19.4%). Those species that tend to be more aggregated also had a higher representation within the PS network. For example, while respectively 65% and 70% of the national total of dark-bellied brent goose and pintail occurred on sites with large numbers, a significant proportion of the total numbers of birds included in the remaining size groups was also protected, including the smaller size categories (Fig. 2.3b and c). Specifically, for dark-bellied brent goose, 80.7% of the national total for sites with <= 2000 birds (0-10% range) and all individuals from the larger size classes were counted on PS wetlands (Fig. 2.3b). For pintail 65.6% of the 0-10% range was recorded on PS (Fig. 2.3c).

Considering only the overall PS and LS networks in 1998/99, again for some species those protected sites containing small numbers of individuals contained large numbers in aggregate (e.g. cormorant, mallard, shoveler, and tufted duck). Although the PS totals for these species were substantial, few of the sites are listed as nationally important for the species and therefore do not contribute to their LS suite. For mallard, none of the sites with a 5-year mean population of <2,000 birds within a $\geq$20,000 assemblage qualify as SPAs for that species (i.e. for its LS suite). Collectively, however, such sites (0-50% range) held 89.9% of the national total and 79.1% of the SPA total of mallard in 1998/99 (Fig. 2.4a). Similarly, for the tufted duck no site with <1% (600 birds) of the national total is eligible for listing (0-10% range), yet these sites contained 70.4% of the national total and 39.5% of the PS total in 1998/99 (Fig. 2.4b). For the coot, 66.1% of the national total and 41.2% of the PS total was recorded on sites outside the coot LS suite, where the total fell below the threshold for listing as sites of national importance in 1998/99 (1,100 birds) (Fig. 2.4c). In addition, for goldeneye, although sites in the 0-10% range collectively accounted for 71.5% of the national total numbers of birds, only three of these sites form part of the LS for this species (Fig. 2.4d).

Table 2.3: The current number of LS (as of 1998/99); the numbers of LS classified using selection criteria 1.1, 1.2, 1.3, and 1.4; and the biogeographic population, national, PS and LS totals for 21 species of waterbird. ([a]no biogeographic population estimate; [b]no LS suite).

Species	Current Number of LS	Numbers of LS classified using criteria Stages: 1.1	1.2	1.3	1.4	Biogeographic population	National total	PS total	LS total
Little grebe	7			7		[a]	3695	933	548
Great crested grebe	10			10		150000	8908	2238	1193
Cormorant	23			23		120000	14597	4492	2866
Bewick's swan	12	12				17000	7169	5870	5449
Whooper swan	12	12				16000	3838	2419	2195
European white-fronted goose	7			6	1	600000	5595	4988	4006
Dark-bellied brent goose	18		6	12		300000	95851	83010	73719
Shelduck	22		9	13		300000	72536	53180	49613
Wigeon	34		5	27	2	1250000	372636	260577	190608
Gadwall	12		5	7		30000	12295	3082	2237
Teal	22		8	14		400000	132493	68797	53128
Mallard	7			7		2000000	150788	44946	16065
Pintail	21		16	5		60000	25617	18780	17586
Shoveler	17		7	10		40000	10187	3705	2686
Pochard	8		2	6		350000	41472	13190	8889
Tufted duck	4		1	3		1000000	53107	8139	5617
Goldeneye	6			6		300000	16736	3604	1299
Smew	0					25000	309	52	[b]
Red-breasted merganser	6			6		125000	4441	1603	614
Goosander	0					200000	4308	595	[b]
Coot	3			3		1500000	107736	20283	12791

Proportions of a species' biogeographic population and the national and PS totals within the PS and LS networks.

Protected Site totals

Calculating the percentage of a species' biogeographic population and the percentage of the national total within the PS and LS networks, based on the most recent 5-year peak mean data (1994/95-98/99), provides an indication of the relative effectiveness of the current suite of PS at both an international and national level. These five-year means are traditionally used to assess site importance, as a way of damping fluctuations in count data (Pollitt et al. 2000, Ramsar Convention Bureau 2002b). When considering the proportion of a species' biogeographic population within the PS and LS networks it should be noted that for each species there is an upper limit, dependent upon the fraction of this population which winters in Great Britain, which varies considerably. Indeed, for the 5-year period 1994/95 to 1998/99 the percentage of the biogeographic population recorded on the British WeBS sites for the 21 species ranged from 0.93% for European white-fronted goose to 42.70% for pintail (Table 2.3).

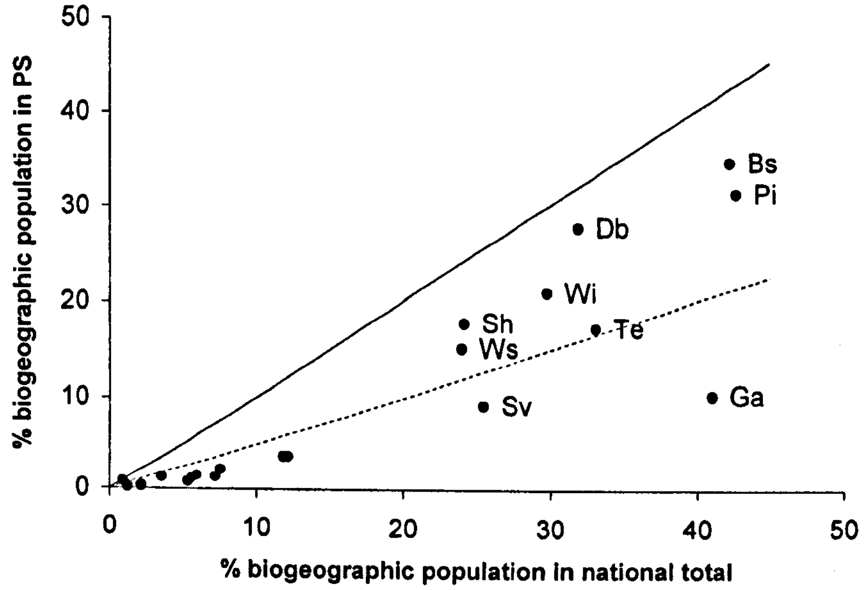

Figure 2.5: The variability in representation for each of the 21 species within the PS network in relation to the proportion of the biogeographic population that occurs in Great Britain. From left to right the first eleven data points refer to species Ew, Sm, Gs, Rm, Td, Ge, Gg, Co, Ma, Po, Cr. The solid line refers to the maximum possible level of representation, the dashed line, 50% of the national total. The symbols are: Gg) great crested grebe, Cr) cormorant, Bs) Bewick's swan, Ws) whooper swan, Ew) European white-fronted goose, Db) dark-bellied brent goose, Sh) shelduck, Wi) wigeon, Ga) gadwall, Te) teal, Ma) mallard, Pi) pintail, Sv) shoveler, Po) pochard, Td) tufted duck, Ge) goldeneye, Sm) smew, Rm) red-breasted merganser, Gs) goosander and Co) coot. (No biogeographic population estimate for little grebe (Lg).

The percentage of the biogeographic population within the PS network was small for many species (Fig. 2.5). For example, <5% of the biogeographic population was included in the network for 11 species (great crested grebe, cormorant, European white-fronted goose, mallard, pochard, tufted duck, goldeneye, smew, goosander, red-breasted merganser and coot). However, in general where the percentage of the biogeographic population recorded in Great Britain was larger, the percentage within the PS was also larger ($r = 0.90$; *d.f.* = 20; $P < 0.0001$). However, there were exceptions to this trend, for example, shoveler and gadwall have >25% of their respective biogeographic populations in Great Britain, but <50% of the total numbers in Great Britain are in the PS network.

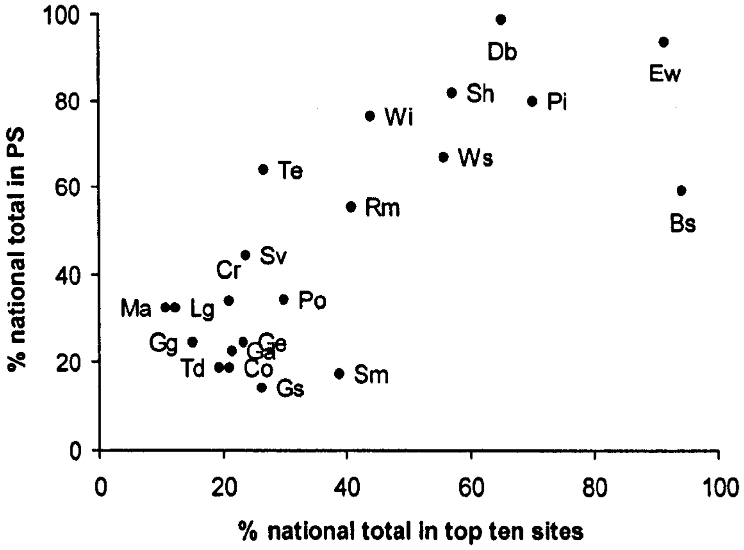

Figure 2.6: The effect of aggregation and dispersal on the level of representation of each species within the PS network in terms of the percentage of the national total that occurs in the top ten WeBS sites for population size (these are not necessarily all PS) and the percentage of the national total found in the PS network. (For key to symbols see Fig. 2.5).

Considering the percentage of a species' national total within the PS network, there was considerable variation between the 21 species analysed, from 13.8% for goosander to 89.2% for European white-fronted goose. Additionally, there was a positive relationship ($r = 0.77$; *d.f.* = 20; $P < 0.0001$) between the percentage of a species' national total within the PS and the percentage of the national total in the top ten sites for that species, whether these sites were in the PS network or not (Fig. 2.6). For those species with a high proportion of the national total within the top ten sites for that species, such as European white-fronted goose, dark-bellied brent goose, Bewick's swan, whooper swan, shelduck and pintail, the PS network supported a greater percentage of the national total (Fig. 2.6). Specifically, >80% of the national population was recorded in the PS network for Bewick's swan (81.9%), dark-bellied brent goose (86.6%)

and European white fronted goose (89.2%). In addition, for a further eight species the network supported >50% (Table 2.3). Conversely, for those species with a more dispersed distribution, the SPA network supported a lower proportion of the national total. Indeed, for nine species, less than 30% of the national total (little grebe (25.3%), great crested grebe (25.1%), gadwall (25.1%), mallard (29.8%), tufted duck (15.3%), goldeneye (21.5%), smew (16.8%), goosander (13.8%) and coot (18.8%), was supported by PS wetlands.

Listed Site totals

For the 21 species, the percentage of the biogeographic population within LS ranged from 0.4% for goldeneye to 32.1% for Bewick's swan, with a mean (±SD) of 9.4% (± 10.6). For seven species (great crested grebe, European white-fronted goose, mallard, tufted duck, goldeneye, red-breasted merganser and coot) the total percentage of the biogeographic population, using the 1994/95-1998/99 peak means, in all LS was <1% (one of the established classification thresholds applied to individual site populations). This contrasts with only four species (little grebe, European white-fronted goose, mallard, coot) with <1% using the 1991/92-1995/96 peak mean data (the data used for the recent SPA review; JNCC 2001).

The percentage of the national total of a species within the LS is inevitably greater than that for the biogeographic population. Indeed, this percentage ranged from 7.8 % for the goldeneye to 76.9 % for the dark-bellied brent goose, with a mean (± SD) of 35.7% (± 26.2) (Table 2.3). In addition, the LS for nine of the species contained <20% of the national total.

The percentage of the PS total within the LS ranged from 35.7% for mallard to 93.6% for pintail, with an overall mean (± SD) of 69.5% (±18.8). For 16 of the 21 species the percentage of the PS total within the LS was >50%, however for three the percentage was <40% (35.7% for mallard, 36.1% for goldeneye and 38.8% for red-breasted merganser) (calculated from Table 2.3). Additionally, there was a significant positive correlation between the percentage of the biogeographic population of a species in the national total and the percentage of the PS total in the LS ($r = 0.69$; $N = 18$; $P < 0.05$). This means that, for those species such as Bewick's swan, gadwall, teal and pintail where a greater percentage of the biogeographic population resides in Great Britain, the percentage of the PS total within the LS was correspondingly greater.

Concordance analysis

There was a high level of concordance between the peaks in numbers across all species when analysed together ($w = 0.71$; $N = 1962$; $P < 0.0001$). Similarly, for all species individually, there was also a significant degree of concordance across all years. The lowest values were for smew ($w = 0.41$; $N = 1962$; $P < 0.0001$) and the highest for dark-bellied brent goose ($w = 0.77$; $N = 1962$; $P < 0.0001$). Ten species had values greater than 0.70 with a further nine greater than 0.50 (Table 2.4).

Table 2.4: The degree of concordance between the peaks of abundance on individual wetland sites for the 21 waterbird species analysed, both in combination (all 21 species) and for each individually (±SD). *$P < 0.0001$

Species	w (±SD)
All 21 species	0.71 (0.04)*
Little grebe	0.63 (0.09)*
Great crested grebe	0.70 (0.10)*
Cormorant	0.66 (0.11)*
Bewick's swan	0.50 (0.09)*
Whooper swan	0.55 (0.08)*
European white-fronted goose	0.43 (0.14)*
Dark-bellied brent goose	0.77 (0.10)*
Shelduck	0.66 (0.07)*
Wigeon	0.72 (0.03)*
Gadwall	0.67 (0.07)*
Teal	0.73 (0.04)*
Mallard	0.75 (0.03)*
Pintail	0.64 (0.04)*
Shoveler	0.74 (0.04)*
Pochard	0.74 (0.04)*
Tufted duck	0.75 (0.03)*
Goldeneye	0.69 (0.03)*
Smew	0.41 (0.08)*
Red-breasted merganser	0.62 (0.09)*
Goosander	0.60 (0.05)*
Coot	0.74 (0.08)*

2.4 Discussion

There have been relatively few in-depth, peer-reviewed evaluations of existing protected area network performance (although see Table 1.4 for examples). The paucity of such reviews exists because they are costly and difficult (Clark 1996), they may provide unwanted suggestions (Kleiman et al. 2000, Kleiman 2003; see also Chapter 10), and their recommendations may be difficult and costly to implement. Where they have been conducted, they typically report insufficient coverage, sub-optimal species/habitat representation and inadequate selection methods (see for example Williams et al. 1996, Khan et al. 1997, Freitag et al. 1998, Nantel et al. 1998). Indeed, some authors have suggested that the evaluated sites appear to be *ad hoc* collections of areas rather than systematically selected reserve networks (see for example Pressey 1994, Pressey et al. 1994, Lombard et al. 1995, Freitag et al. 1998) and are, as a result, inadequate for the protection of target species (Rebelo and Siegfried 1992). Few studies report the overall success of reserve networks; notable exceptions being for fenland SSSIs in Scotland (Rodrigues et al. 1999) and RSPB reserves in Great Britain (Hopkinson et al. 2000b). Nevertheless, in contrast to the many negative reports of reserve network performance, the network of SPAs and Ramsar Sites in Great Britain has been shown to be extremely successful in terms of the overall numbers of waterbirds regularly supported. Indeed, considering all WeBS species, despite accounting for <3% of the total number of wetland sites, those sites with large overall waterbird numbers (≥20,000 birds) support approximately two-thirds of the total numbers of wintering waterbirds. Bearing in mind the extreme pressure on the classification and designation process, both from competing land use options and from the European Union to meet set quotas and targets, this level of protection is a remarkable achievement, particularly in comparison with other countries (e.g. Table 1.4). For example, the Ramsar network in Mexico was strongly criticised by Perez-Arteaga et al. (2002), who concluded that the seven existing Ramsar sites are not representative of the diversity of Mexican wetlands. Furthermore, the current number of Ramsar Sites officially listed on the 'IUCN Protected areas of the world database' for Great Britain (140) greatly exceeds that of all other signatories (Australia 63, Canada 36, France 18, Greece 10, Mexico 7, Portugal, 12, South Africa 17 and USA 19) and provides an indication of the considerable effort made to adopt the targets set out both in the Birds Directive and the Ramsar Convention.

In addition to this excellent performance in total conservation, for three species (Bewick's swan, European white-fronted goose and dark-bellied brent goose) >80% of the national population was recorded within PS, and >50% for eight additional species. These species generally are those where the proportion of the biogeographic population in Great Britain is greatest (with the exception of European white-fronted goose which has only 0.9% of the biogeographic population in Great Britain), suggesting that the SPA/Ramsar Site selection process has successfully preferentially targeted those species of international importance over others. It should be borne in mind, however, that these proportions of the biogeographic population are likely to be an over estimate given that not all wetlands in Great Britain are included by WeBS (see Chapter 1 for a discussion of the limitations of WeBS).

For several species, however, substantial total numbers of birds occur in low numbers on many sites (e.g. mallard, tufted duck, goldeneye and coot). Further, their combined value exceeds that of sites that individually contain larger numbers of individuals. Nonetheless, such sites will not generally be classified as SPAs/Ramsar Sites following the current site-selection methodology (see also Chapter 3.1). Given that the 1% threshold levels to determine the national and international importance of a site were devised preferentially to target aggregating species (Atkinson-Willes et al. 1982) such as Bewick's swan, whooper swan, European white-fronted goose, dark-bellied brent goose, shelduck and pintail this was not unexpected. Indeed, the PS network better represents these aggregating species as the numbers of birds on a site regularly exceed the numerical site-selection thresholds. In consequence, there are considerable differences between species in the proportion of the national total found within the PS network, which is largely a function of their national and international distribution across wetland sites.

Although protection of the sites with highest waterbird numbers means that high numbers of birds are protected with minimum site inclusion, as shown by the strongly aggregated nature of wintering waterbirds in total (i.e. considering all WeBS species together), a significant proportion of the national total remains unprotected and there is a bias towards species that tend to occur in the most populous sites. This assumes that the most important sites for target species, and therefore a sufficient proportion of each of their overall populations, occurs within these larger assemblages. If a significant proportion of a species' population is found elsewhere, alternative approaches may be more appropriate to ensure adequate protection. Indeed, the degree of aggregation does not necessarily reflect conservation priority. In particular, those species that are widespread and abundant today may be in great need of conservation in the future, as evidenced by the recent declines of a number of once common bird species characteristic of farmland habitats (Fuller 2000, Gates and Donald 2000). For example, although the mallard is widespread and has a large population size, it is also one of the few waterbird species where the numbers recorded in Great Britain over winter have declined in recent years (at a rate of 3% per year, 1989-99; M. Kershaw, unpublished analyses). Furthermore, several dispersed distribution species including shoveler, pochard and goldeneye were included as species of conservation concern in the JNCC 'Birds of Conservation Concern 2002-2007' (JNCC 2002). Specifically, for shoveler, internationally important numbers regularly winter in Great Britain (>25% of the biogeographic population), nonetheless, protected areas support <50% of the national total. By contrast, Great Britain supports <1% of the biogeographic population of European white-fronted goose, yet almost 90% of the national total winters on protected areas.

A species' presence in the PS network does not guarantee protective measures specific to that species as only individuals on LS are targeted for active management. However, the percentage of a species' population on such sites is even smaller than on the PS network, and for many species is biased towards those SPAs that hold large numbers of waterbirds overall, rather than the most important sites for that species. The relatively low percentages of the PS total within some species' LS suggest, as is intuitively expected, that the listing of LS using the current site-selection criteria does not inevitably target the most suitable sites for protection (see Chapter 3.1 for further discussion). For those species where the population does not aggregate and are thus not the focus of the SPA selection process, the most important sites on which they occur will not necessarily hold >1% of the biogeographic population or ≥20,000 waterbirds, as is in fact the case for several of these species. If this is so, then many potentially important wetland sites for these species will be ignored by the selection guidelines (see Chapter 3.1). As an example, the pochard LS suite contains 11 sites in Great Britain, but four of the top ranking sites for pochard, based on the 5-year peak means for 1994/95 to 1998/99, are not included in the LS because they do not contain ≥20,000 waterbirds. Additionally, 34 LS have been listed as important for wigeon, a species which occurs in large numbers on many wetland sites, compared with only four for tufted duck and seven for mallard, which are more dispersed in distribution and do not occur in sufficient numbers in the larger (≥20,000) assemblages. Nonetheless, for internationally important species (i.e. where there is a large proportion of the biogeographic population within Great Britain) there tends to be a greater percentage of the population within the LS.

Site fidelity is an important aspect of the life cycle of many species of migratory organisms, for example, fur seals (Baker et al. 1995), songbirds (Warkentin and Hernández 1996) and various waterbirds (Nickell 1968, Nisbet and Medway 1972, Berthold 1993, Warkentin and Hernández 1996, Lawton 2000). In accordance with these authors, each of the 21 species analysed here exhibits a statistically significant degree of concordance between the peaks of abundance in consecutive years, indicative of wintering site fidelity. In conservation and management terms, it is critical for the continued existence of a species that historically significant/preferred sites are included in a protected area network (i.e. as part of the PS). This is particularly important for those species with a dispersed distribution across suitable wintering sites given that many important sites will not inevitably be identified as potential SPAs or Ramsar Sites using the 1% threshold approach. However, because strong fidelity to individual sites may mean that a species is less able to adapt to habitat degradation and loss (Warkentin and Hernández 1996), it is insufficient to simply recognise their importance and to stop at the identification phase. Inevitably, careful management, to maintain the integrity of a site and to ensure that it continues to be able to support viable populations, is of paramount importance if a species is to persist (see Chapter 7).

2.5 Conclusions

In sum, the performance of the current PS network is remarkable, particularly in comparison with published analyses of networks elsewhere in the world (Table 1.4). Nonetheless, the current site-based approach, whilst having the great benefit of simplicity, is deliberately biased towards aggregating species at the expense of the more dispersed distribution species such as mallard, shoveler, pochard, goldeneye and goosander, many of which are species of conservation concern (JNCC 2002). To achieve an appropriate level of representation for all target species (as defined by the Birds Directive) will require further development of the current site-selection criteria. Specifically, to ensure that the PS network continues successfully to protect nationally and internationally important waterbird populations, efforts now need to concentrate on the derivation of species-specific representation targets, and in particular, the ways in which these can be incorporated into the site selection process (discussed in detail in Chapters 5-7).

CHAPTER 3: NETWORK PERFORMANCE

'It is not the label that matters but what the labels achieve (Marren, 2002).'

3.1 SPA and Ramsar sites:

The performance of protected areas for migratory waterbirds.

3.1.1 Introduction

Wetlands in Great Britain are of considerable value both on a national and an international scale, in part because of the substantial numbers and diversity of migratory waterbird species reliant on them for breeding, feeding, roosting and resting at various times during the year. Almost inevitably however, given the wide variety of ecosystem goods and services they provide (Table 3.1.1), wetland habitat losses due to human exploitation have been extensive (Moore et al. 1989, Lindegarth and Chapman 2001). For example, approximately 88% of estuaries in Great Britain were affected by land reclamation alone in 2000 (Austin et al. 2000).

Table 3.1.1: Direct and indirect use values of wetlands.

Direct Use values	Indirect Use Values
Construction materials	Flood control
Fishing materials	Groundwater recharge
Food and drink	Shoreline stabilisation
Medicines	Storm protection
Recreation and tourism	Water quality improvement
Transport	(Micro) climate change mitigation
Agriculture	Nutrient removal and transformation
Fuel/Energy	Sediment and toxin retention
	Aquatic diversity and abundance
	Wildlife diversity and abundance

In response, there has been an increasing emphasis on the conservation and sustainable use of wetlands, principally through the design and implementation of international conservation agreements. Indeed, wetlands are one of the few ecosystems to be the sole focus of an international conservation convention, namely the Ramsar Convention (Ramsar Convention on Wetlands of International Importance Especially as Waterfowl Habitat); the first of the intergovernmental treaties devoted to the conservation and wise use of natural resources.

In direct contrast to the apparent lack of foresight regarding many historic protected area designations (Margules et al. 2002), in the UK the selection of nationally important sites as Special Protection Areas (SPAs) and those of international importance as Ramsar Sites for waterbirds is achieved through the application of numerical threshold guidelines pertaining to the numbers of individuals at a given site (discussed in detail in Chapter 2). Within the Birds Directive in addition to the detailed site-selection requirements there are in place conditions regarding the eventual management of selected sites. Specifically, under international law Member States are required to take the requisite measures to ensure that populations of target species listed in Article 1 are maintained at 'a level which corresponds in particular to ecological, scientific and cultural requirements, while taking account of economic and recreational requirements, or to adapt the population of these species to that level' (Article 2). In addition, 'application of the measures taken pursuant to this Directive may not lead to deterioration in the present situation as regards the conservation of species of birds referred to in Article 1'. In this respect, each state must submit a report to the European Commission every three years, detailing progress and status to-date (Article 12).

In light of the EU legislation and the Ramsar Convention recommendation for management at the local site level and the current numerical thresholds approach to site-selection, the expectation is that sites identified as part of the SPA/Ramsar network (the PS network) in the UK would contain greater numbers of birds compared with non-PS wetland sites, greater numbers than expected by chance (a common bench mark, e.g. Hopkinson et al. 2000b, Virolainen et al. 2000, Williams et al. 2000, Brooks et al. 2001, Bonn et al. 2002) and would be essentially buffered with respect to national-level changes. In this Chapter I test these assumptions for a selection of waterbirds common to wetland sites in the UK, to assess the performance of the PS network for these species.

3.1.2 Methods

Twenty-one species of waterbird were analysed separately. Data for the years 1975/76 (first PS designated) to 1998/99 (latest available data as of September 2001) inclusive were included, for little grebe (data only available from 1985/86), great crested grebe (data only available from 1982/83), cormorant (data only available from 1986/87), Bewick's swan, whooper swan, European white-fronted goose, dark-bellied brent goose, shelduck, wigeon, gadwall, teal, mallard, pintail, shoveler, pochard, tufted duck, goldeneye, smew, red-breasted merganser, goosander and coot (data only available from 1982/83). These 21 species are well represented by WeBS counts, for which these data are the principal sources used for the selection of SPAs and Ramsar Sites in the UK.

National totals

For each of the 21 species individually, following the procedure established by WeBS (Pollitt et al. 2000), the national total (numbers of birds recorded nationally) for a given month was take to be the sum of the individual monthly counts across the 1962 selected WeBS sites in Great Britain (see Chapter 1). The peak of these monthly counts in each year was taken to represent the national total for each species in each year (1975/76-1998/99 unless previously stated).

Individual site totals

For the 21 species separately, counts from each WeBS count unit were summed to provide estimates for each site. The maximum number of individuals recorded on each of these sites between September and March was then determined for each year 1975/76-1998/99 (peak annual site count). Annual totals for all PS (up to and including those classified/designated in 1998/99), taken as the sum of the individual site counts, were calculated for each count year from 1975/76 to 1998/99. In addition, for each species, totals for all sites listed as nationally important (sites supporting ≥ 1% national population of a species or ≥ 2,000 birds) were summed to provide annual counts for the LS suite.

The PS network is not static and new sites are classified/designated in each year (see Fig. 2.1). These analyses (unless otherwise stated) were, therefore, carried out assuming that the current (1998/99) PS network and LS suite had existed since 1975/76. The annual totals for each species, therefore, represent the back-calculated totals for this set of sites from 1975/76-1998/99. This means that SPAs and Ramsar sites identified after this date will not be included. It should be borne in mind that the PS network in Great Britain is constantly evolving and now includes additional sites, but given that data were not available these have not been included here.

Analyses

Using the five-year peak mean data for the period 1994/95-1998/99 (five year peak means are currently used by WeBS to represent population sizes), 10,000 random site combinations, each without replacement, were generated and the total numbers of individuals in these hypothetical networks was compared with the numbers in the actual PS network or LS suite. Each species' random network comprised a number of sites equal to that in the current (1998/99) PS network (N =138 in all cases) or LS suite for that species (the number of sites varies for each individual species; see Table 2.3). As an example, the LS suite for cormorant comprised 28 sites. Thus, for each comparison, 10,000 random networks of 28 sites were generated, selected without replacement from all sites. Considering LS suite comparisons, only those species for which there are official LS were included, thereby excluding smew and goosander.

Temporal correlations between the national total and the proportion of this total within the PS network for each species were calculated to provide an indication of the degree to which fluctuations in the national total were reflected in the PS network. Considering each species individually, these correlations were based on count years from the first record of a PS total (1975/76 for the majority of species) to 1998/99 and also separately between 1990/91-1998/99 to reflect recent trends. In all cases the PS network was assumed to have existed since 1975/76.

3.1.3 Results

Temporal trends in numbers

The total numbers of individuals in Great Britain (national total) increased significantly between 1975/76-1998/99 (Table 3.1.1) for 19 of the 21 species analysed. For example, numbers of gadwall increased dramatically from 1,063 birds in 1975/76, to 15,197 in 1998/99 (Fig. 3.1.1a). Similarly, numbers of wigeon increased from 118,669 birds in 1975/76 to 367,873 birds in 1998/99. However, for mallard, although there was a slight increase in numbers nationally, this was not significant (Table 3.1.1). By contrast, for European white-fronted goose there was a slight decrease in numbers nationally, this was, however, not significant. For six species (whooper swan, pintail, pochard, smew, goosander and red-breasted merganser), despite the significant increase in numbers, the national total fluctuated markedly between 1975/76 and 1998/99 (e.g. pintail; Fig. 3.1.1b). In contrast to the increase in numbers nationally between 1975/76-1998/99 for each of these species, in more recent years (1990/91-1998/99) five species actually decreased (great crested grebe, Bewick's swan, dark-bellied brent goose, shelduck and mallard; Table 3.1.1). The declines were, however, significant for only two of these (dark-bellied brent goose and mallard). Specifically, numbers of mallard wintering in Great Britain decreased from 206,229 birds in 1990/91 to 150,216 birds in 1998/99, an overall decline of approximately 27% (Fig. 3.1.1c). By contrast, the national trend was positive for the remaining 16 species, although this was significant for only four of these (little grebe, gadwall, tufted duck and coot; Table 3.1.1). The SPA and Ramsar networks are continually evolving, with additional sites classified/designated in each year. Accordingly, the expectation was that the numbers of individuals of each of the species supported by the PS and LS would exhibit a corresponding increase. Indeed, for each of the species analysed, the PS and LS totals increased significantly between 1975/76-1998/99 (Table 3.1.1).

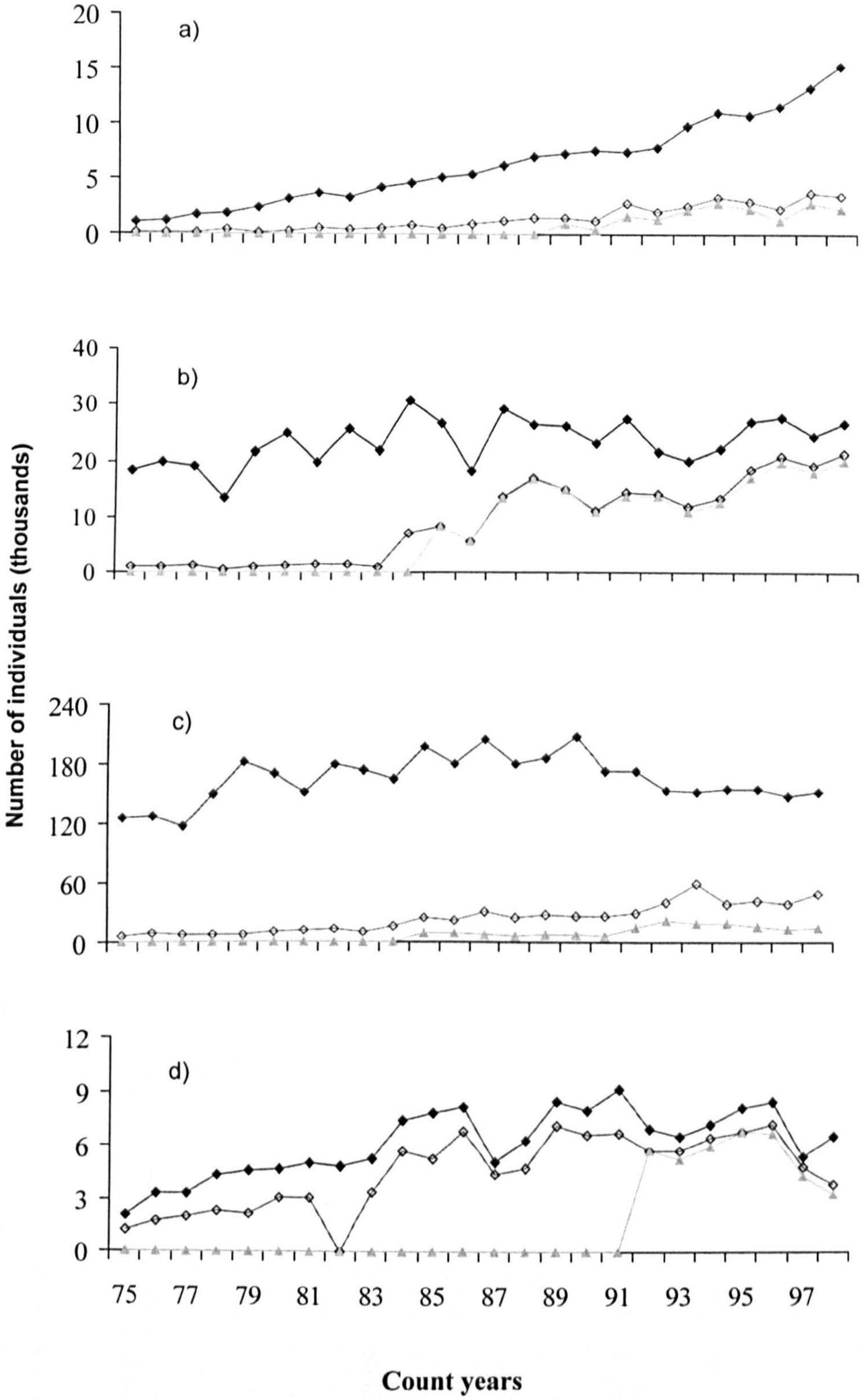

Figure 3.1.1: The national (filled diamonds), PS (open diamonds) and LS (triangles) totals calculated from 1975/76 to 1998/99 for: a) gadwall, b) pintail, c) mallard and d) Bewick's swan.

Table 3.1.1: Temporal trends (slopes (±SE) reported) in the total numbers of individuals nationally, for PS ($N = 138$) and for LS (for N see Table 2.3) calculated for the periods 1975/76-1998/99 (unless otherwise stated in the methods) and 1990/91-1998/99. * $P < 0.05$, **$P < 0.01$, ***$P < 0.001$. [a]No LS.

	1975/76-1998/99			**1990/91-1998/99**		
Species	**National**	**PS**	**LS**	**National**	**PS**	**LS**
Little grebe	241.1 (31.0)***	86.3 (9.7)***	55.1 (7.3)***	152.8 (49.0)*	116.9 (15.7)**	70.3 (16.3)**
Great crested grebe	229.9 (34.6)***	152.5 (34.6)***	140.6 (15.6)***	-9.3 (67.0)	155.0 (53.3)*	104.2 (45.7)
Cormorant	708.2 (167.2)**	380.2 (32.5)***	242.1(36.7)***	98.7 (134.8)	375.7 (70.1)**	173.8 (7.0)*
Whooper swan	103.1 (18.9)***	127.0 (7.1)***	121.1 (11.6)***	0.1 (52.8)	141.9 (20.8)***	156.4 (42.6)**
Bewick's swan	197.4 (39.3)***	227.4 (42.9)***	275.5 (53.1)***	-199.6 (138.2)	-211.4 (127.2)	493.9 (316.6)
European white-fronted goose	-20.1 (40.3)	276.4 (28.1)***	199.8 (35.4)***	55.9 (204.2)	233.3 (189.8)	491.3 (185.6)*
Dark-bellied brent goose	2684.4 (509.2)***	4437.1 (363.9)***	4000.4 (465.4)***	-4831.0 (1076.0)*	4039.2 (1593.0)*	570.7 (2159.0)*
Shelduck	1081.2 (2649.0)***	1859.3 (615.7)**	2639.1 (264.1)***	-788.8 (855.7)	4897.3 (839.7)**	4901.0 (613.9)***
Wigeon	10827.0 (1078.0)***	12008.0 (1416.0)***	9256.0 (1342.0)***	9928.7 (4742.0)	21764 (7745.0)	28531.4 (3693.0)***
Gadwall	549.9 (25.9)***	153.9 (12.8)***	120.1 (17.0)***	947.4 (96.4)***	217.3 (74.6)*	172.6 (86.2)
Teal	3254.6 (345.7)***	1743.1 (318.7)***	1253.7 (288.0)***	1732.8 (1265.0)	7648.5 (988.8)***	6883.7 (552.4)***
Mallard	655.8 (628.6)	1915.5 (172.1)***	902.2 (103.3)***	-5531.0 (1578.0)**	2409.8 (1158.0)	783.2 (629.6)
Pintail	285.5 (110.3)***	982.3 (82.2)***	1002.0 (94.6)***	415.3 (363.3)	1264.0 (258.2)**	1137.1 (257.1)**
Shoveler	183.0 (29.3)***	166.4 (12.3)***	149.8 (12.3)***	79.1 (199.5)	235.0 (67.1)**	176.8 (58.4)*
Pochard	264.3 (106.1)*	542.5 (73.3)***	460.0 (78.9)***	463.6 (378.1)	870.7 (405.0)	821.2 (470.8)
Tufted duck	640.4 (101.4)***	380.0 (35.6)***	301.2 (43.4)***	612.1 (215.8)*	737.8 (143.3)**	607.4 (163.5)***
Goldeneye	425.7 (35.3)***	176.4 (12.0)***	71.4 (10.6)***	74.6 (105.0)***	314.4 (29.0)***	121.7 (46.7)*
Smew	10.4 (3.0)**	2.1 (0.5)**	[a]	30.9 (13.5)	8.0 (2.8)	[a]
Red-breasted merganser	115.2 (25.0)***	81.3 (7.9)***	32.5 (3.2)***	59.3 (103.3)	173.8 (28.7)**	62.2 (10.2)**

Goosander	108.1 (28.7)***	30.3 (2.8)***	[a]	196.8 (156.7)	43.3 (17.5)*	[a]
Coot	1948.6 (287.2)***	767.4 (202.7)**	1095.2 (261.0)**	3169.7 (699.8)**	375.5 (729.3)	311.0 (850.7)

Considering the PS total between 1990/91-1998/99, 20 species showed an increase in numbers; of which 15 were significant. Conversely, for Bewick's swan, the PS total actually decreased (Table 3.1.1; Fig. 3.1.1d). For example, the numbers of individuals on the PS network decreased from 7,268 birds in 1996/97 to 3,880 birds in 1998/99 (approximately 47%; Fig. 3.1.1d). By contrast, the LS for all species increased. This was, however, not statistically significant for great crested grebe, Bewick's swan, mallard, gadwall, pochard and coot; Table 3.1.1).

Table 3.1.2: The numbers of randomly generated sets of WeBS sites supporting a greater number of individuals than the existing PS network or LS suite for each of the 21 species individually. Significance levels are the one-tailed probabilities given by the number of site combinations where the total is greater in the 10,000 random combinations than for the existing PS network/LS suite (1994/95-1998/99 peak means). *$P < 0.05$; **$P < 0.01$; ***$P < 0.001$, ****$P < 0.0001$. x No LS.

Species	PS	LS
Little Grebe	64**	0****
Great Crested Grebe	222**	0****
Cormorant	4***	1***
Whooper Swan	38**	1***
Bewick's swan	47**	7***
European white-fronted goose	6***	0****
Dark-bellied brent goose	27**	0****
Shelduck	39**	1***
Wigeon	3304	3***
Gadwall	74**	0****
Teal	7**	0****
Mallard	12**	0****
Pintail	148**	3***
Shoveler	5649	0****
Pochard	2***	0****
Tufted duck	331*	0****
Goldeneye	1742*	30**
Smew	2004	x
Red-breasted merganser	33**	2***
Goosander	4418	x
Coot	97**	0****

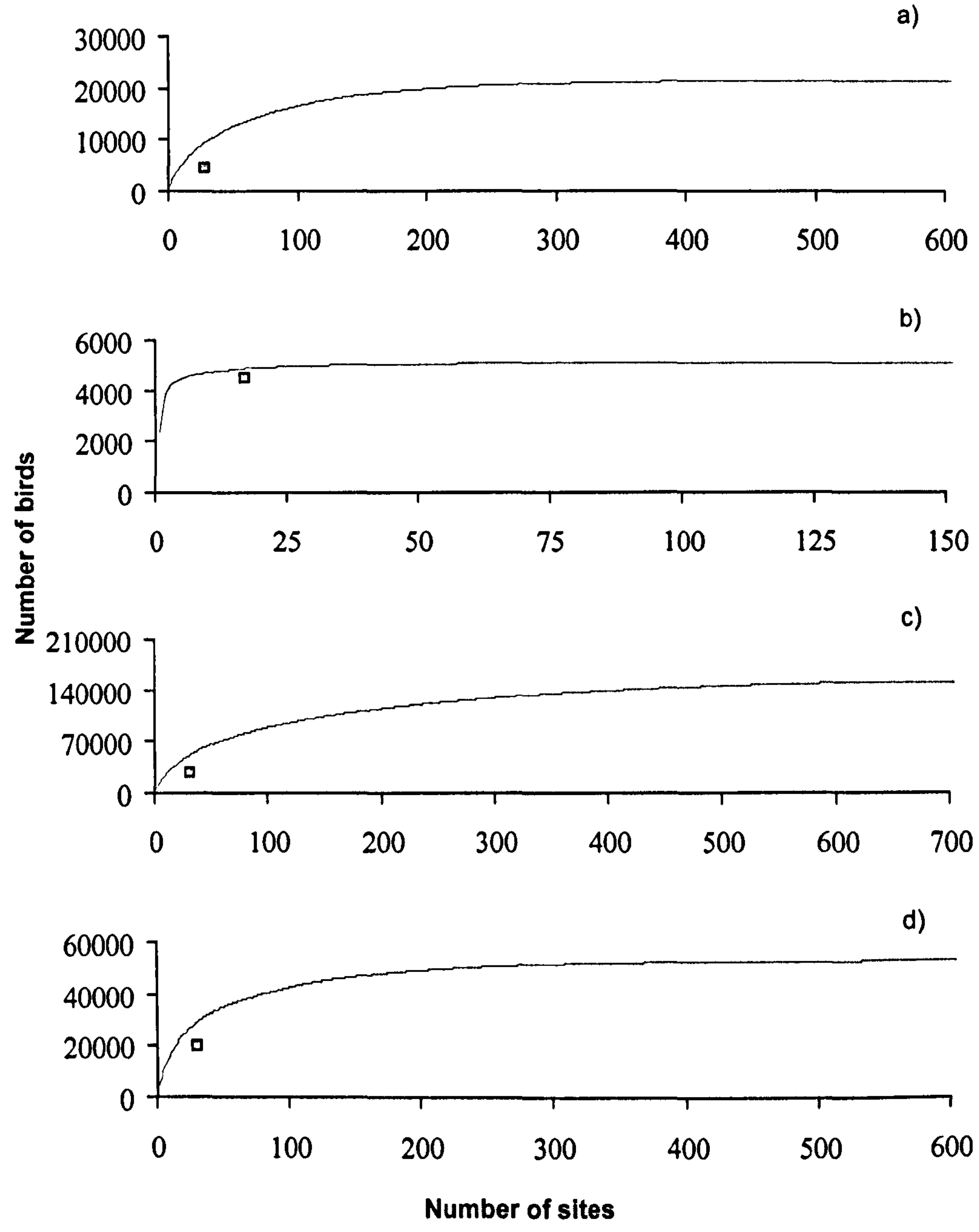

Figure 3.1.2: The cumulative number of birds across all WeBS sites (line) ranked in decreasing numerical order and the total number of birds supported by the PS network (filled square) plotted against the number of sites in each for a) cormorant, b) European white-fronted goose, c) mallard, and d) pochard.

Comparison with random sets

Considering the PS network for each of the 21 species individually (*N* = 138), the total numbers of birds (five-year peak mean 1994/95-1998/99) within these protected sites was greater than expected by chance for 17 species (Table 3.1.2). Further, for ten of these, less than 50 of the 10,000 randomly selected sets of sites contained greater numbers of birds than the existing PS network (Table 3.1.2). This is exemplified by a small deviation from the maximum possible numbers of birds that could be supported in the same number of WeBS sites (WeBS sites ranked in descending numerical order) (Fig. 3.1.2a-d).

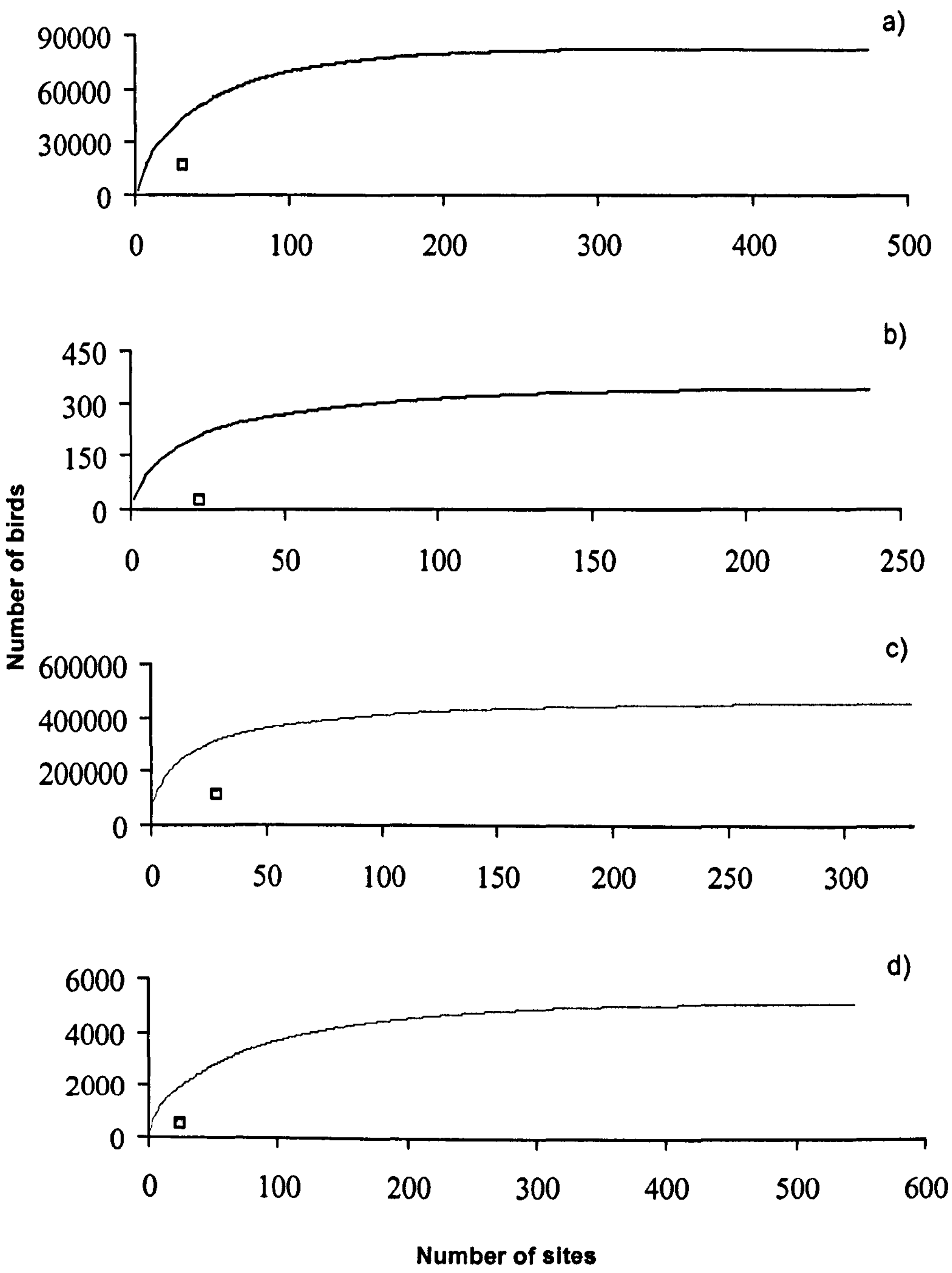

Figure 3.1.3: The cumulative number of birds across all WeBS sites (line) ranked in decreasing numerical order and the total number of birds supported by the PS network (filled square) plotted against the number of sites in each for a) shoveler, b) goosander, c) wigeon and d) smew.

Nonetheless, for all 21 species there was at least one combination that included a greater number of birds than the PS network. For four species (shoveler, wigeon, smew and goosander), the existing PS network contained no more individuals than the randomly selected sets (Table 3.1.2). Specifically, for shoveler, more than half (56%) the randomly selected site combinations contained greater numbers of birds than the actual PS network (Fig. 3.1.3a). Similarly, 44% of the 10,000 random sets for goosander (Fig. 3.1.3b), 33% for wigeon (Fig. 3.1.3c) and 20% for smew (Fig. 3.1.3d) contained greater numbers than the PS network (Table 3.1.2). For each of these four species, the PS network contained considerably fewer birds compared with the potential maximum numbers within the same number of WeBS sites (approximately 41% of the potential numbers of shoveler, 37% of wigeon, 27% of goosander and 13% of smew). For each of the 19 species for which LS have been officially classified (i.e. excluding smew and goosander), the existing LS suite contained significantly higher proportions of a species' national total than expected by chance (Table 3.1.2). Furthermore, for eleven of these species, none of the random sets contained greater numbers than the existing LS and for a further three species, only one combination of sites included more birds (Table 3.1.2). However, for goldeneye, 30 of the randomly selected sets of sites contained greater numbers of individuals than the actual LS (Fig. 3.1.4).

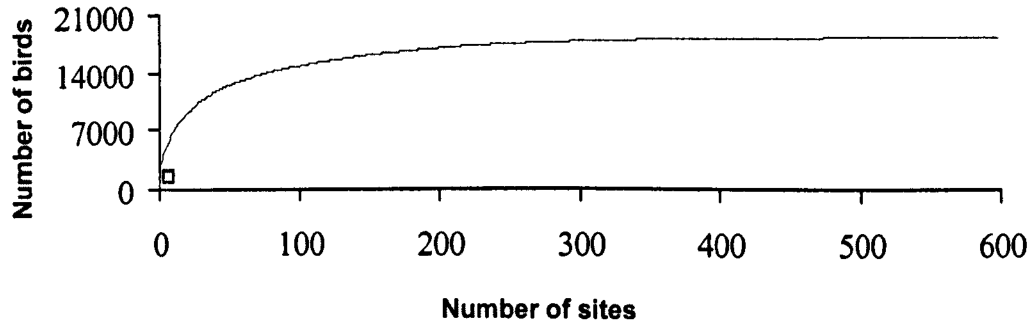

Figure 3.1.4: The cumulative number of birds across all WeBS sites (line) ranked in decreasing numerical order and the total number of birds supported by the LS suite (filled square) plotted against the number of sites in each for goldeneye.

How closely have numbers on the PS tracked national population changes?

Correlation coefficients between the proportion of the national total on the PS ($N = 138$) and the size of the national total in each year from 1975/76 (or earliest available data specified in the Methods section of this chapter) to 1998/99 (based on the current (1998/99) PS network; i.e. assuming that this network had existed since 1975/76) were negative for 14 of the 21 species analysed (Table 3.1.3). While only five of these correlations were significant (cormorant, dark-bellied brent goose, shelduck, wigeon and gadwall), the tendency was for the proportion of birds on the PS to be greater where the national total was lower. Gadwall showed the most significant negative correlation ($r = -0.84$; $P < 0.0001$) and has also experienced the most dramatic increase in national numbers since 1975/76 (Fig. 3.1.5a). Similarly, for cormorant, where the national total was low the proportion within the PS network was high (Fig. 3.1.5b).

Table 3.1.3: Correlation coefficients (Spearman's rank) between the proportion of the national total in the PS network and the national total for each species using data between (A) 1975/76-1998/99 and (B) 1990/91-1998/99. To maintain a constant number of sites the PS network was taken as the suite of SPA and Ramsar sites present in 1998/99. * $P < 0.05$; ** $P < 0.01$; *** $P = 0.001$.

Species	*R* (A)		*R* (B)	
Little grebe	0.06		0.88	**
Great crested grebe	0.55	*	0.36	
Cormorant	-0.90	***	-0.06	
Whooper swan	-0.30		-0.41	
Bewick's swan	0.06		0.25	
European white-fronted goose	0.05		-0.40	
Dark-bellied brent goose	-0.70	***	-0.38	
Shelduck	-0.40	*	0.42	
Wigeon	-0.40	*	-0.02	
Gadwall	-0.84	***	-0.73	*
Teal	-0.10		0.33	
Mallard	0.56		0.06	
Pintail	-0.20		-0.20	
Shoveler	-0.30		-0.51	
Pochard	0.02		0.12	
Tufted duck	-0.50		0.40	
Goldeneye	-0.40		0.02	
Smew	-0.10		0.41	
Red-breasted merganser	0.02		0.89	**
Goosander	-0.10		-0.54	
Coot	-0.20		-0.31	

By contrast, for seven species these correlations were positive (little grebe, great crested grebe, Bewick's swan, European white-fronted goose, mallard, pochard and red-breasted merganser). However, this was statistically significant for great crested grebe alone (Fig. 3.1.5c).

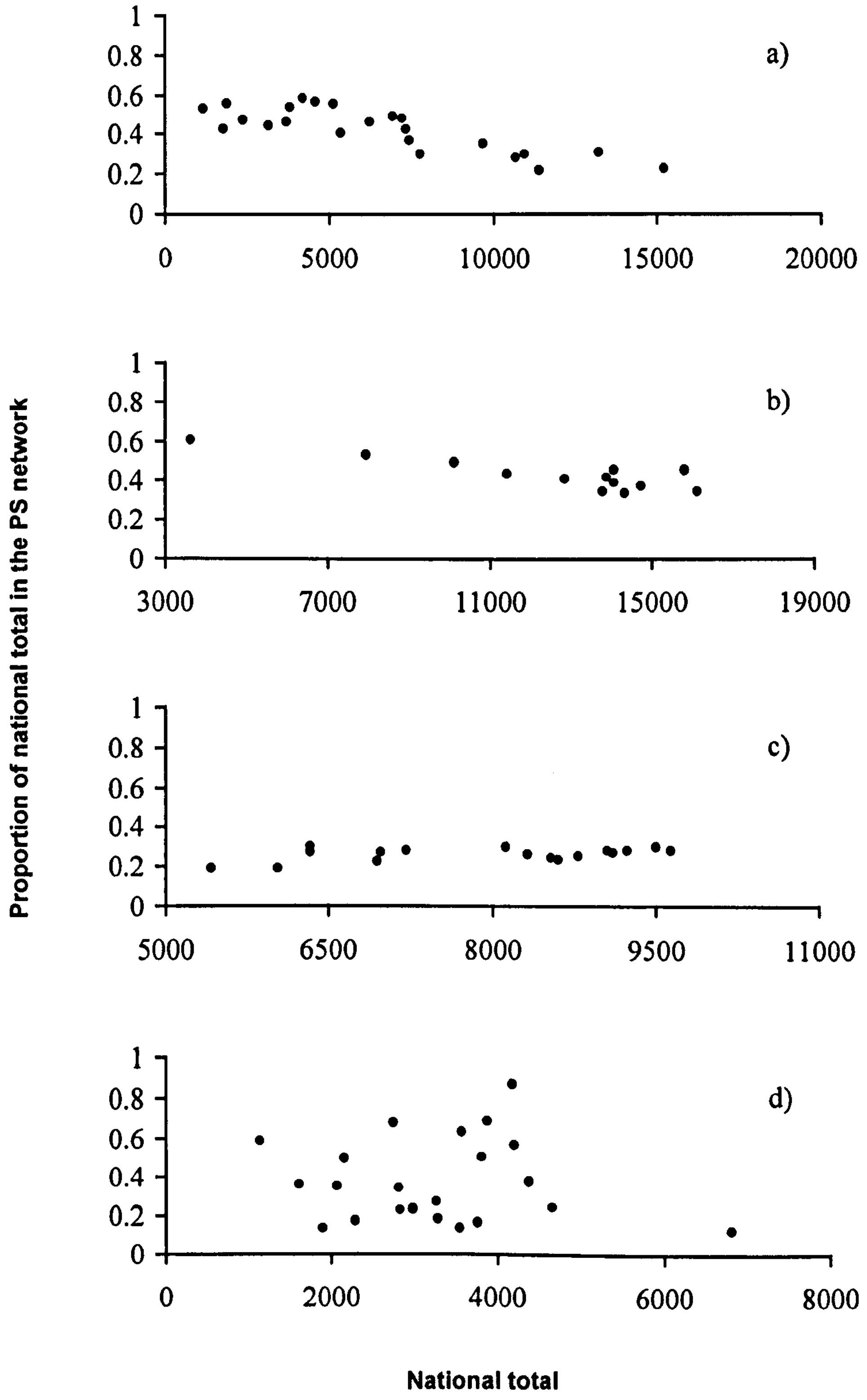

Figure 3.1.5: The relationship between the proportion of the national total within the PS network and the national total between 1975/76 and 1998/99 for: a) gadwall, b) cormorant, c) great crested grebe and d) pochard. Counts based on the current (1998/99) set of PS to avoid problems associated with successive site designations.

Nonetheless, for these species, in general, where the national total was high, the proportion within the PS network was also high. For 15 of the 21 species there was no significant correlation between the national total and the proportion of this total within the PS network. The weakest correlations were for pochard (Fig. 3.1.5d) and red-breasted merganser ($r = 0.02$; $P > 0.05$).

Considering the period 1990/91-1998/99, correlations between the national total and the proportion within the PS network were statistically significant for just three species (little grebe, gadwall and red-breasted merganser) (Table 3.1.3). Specifically, for little grebe and red-breasted merganser the trend was positive, whereas there was a negative relationship for gadwall. The remaining 18 species showed no significant correlation.

3.1.4 Discussion

Random comparisons

The SPA/Ramsar identification and selection process is inevitably time and resource intensive, and once a site is awarded protected area status this cannot, and arguably should not, easily be revoked. It is, therefore, of overriding importance not only that the most suitable sites are chosen in the first place, but also that these are successfully managed so as to maintain viable populations into the future. Clearly, of value to planners, managers and conservationists alike is the persistence of the target species and habitats, rather than directing attention simply towards present-day distributions and abundance (Rodrigues et al. 2000). Not only is this vital in light of the increasing pressure on wetlands for anthropocentric development, which is likely to diminish the opportunities for conservation in the future, but also so that planners and managers can confidently justify a site's inclusion in the network should conflicts concerning constituent sites arise. In this respect, for 17 of the 21 species analysed the PS network did indeed support significantly greater numbers of birds than expected by chance. Although not particularly surprising given the current numerical threshold approach to site selection, it is, nonetheless, highly encouraging, not least for European white-fronted goose, dark-bellied brent goose, gadwall, teal, pintail, pochard and goldeneye, which were individually awarded 'Amber' alert status in the JNCC 'Birds of Conservation Concern 2002-2007' (JNCC 2002). By contrast, for four species (shoveler, wigeon, smew and goosander) the current PS network did not support greater numbers of birds than predicted from 10,000 random sets of the same number of sites. These species typically exhibit a dispersed distribution across their winter range (Cramp and Simmons 1994; see also Chapter 2). Therefore, in view of the generally low numbers of individuals supported by the majority of sites, limited numbers of these areas will qualify as nationally/internationally important under the current numerical threshold approach to SPA/Ramsar Site selection (Chapter 2). Consequently, following the current site-selection procedure, it is not inevitable that all of the best sites for such species will be included in the PS network (Chapter 2). Indeed, the results of these analyses show that, for each of the 21 species individually, the PS network always supported less than the maximum possible numbers of individuals for that number of sites. Under the EU Birds Directive all countries are required to take measures for the protection of all species listed in Annex I/II and not simply those that form dense aggregations. In light of the inadequate level of representation achieved for these four species (shoveler, wigeon, smew, goosander) by the current PS network, it will, therefore, be necessary to re-address the current approach to priority-site selection, to incorporate a greater proportion of the national total at least within the PS network as a whole and ideally within the LS suite for each species individually (discussed in more detail in Chapters 4-6). Indeed, given that two of these species (wigeon and shoveler) are listed as 'Amber alert' species in the JNCC 'Birds of Conservation Concern', an effective network of protected areas should be a priority.

In contrast to the variation in effectiveness shown for the PS network amongst the 21 species, the LS always supported significantly greater numbers of individuals compared with the 10,000 random sets of sites. This is extremely reassuring, particularly given that because the LS suite for a species is identified exclusively from those sites that already qualify as SPAs or Ramsar Sites (i.e. those sites supporting ≥20,000 birds or ≥1% of the biogeographic population/national total), it is not inevitable that all the most populous sites for a single species are those fulfilling the overall PS selection criteria (Chapter 2). In particular, for those species which do not tend to aggregate or which are not generally found at sites where there are >20,000 birds, some of the top sites may be excluded from the LS listing process. Specifically, for goldeneye it is clear that the current selection criteria for the identification of LS are not targeting all sites of numerical importance. This is of particular concern as management plans for individual target species (as defined by the Birds Directive) are biased towards those sites considered nationally important (i.e. the LS suite for a species). In consequence, where some of the best sites for a species are not recognised as such, the long-term protection of the species will inevitably be compromised.

Temporal fluctuations

For the 21 species for which data have been analysed, Great Britain forms only a part of the winter range. Therefore, some of the fluctuations in numbers at national level reflect, or are at least partly attributable to, changes in distribution and abundance occurring elsewhere (Stroud et al. 1990). For example, periods of severe weather elsewhere in Europe alter distributions, and thus the proportion of the biogeographic populations occurring in Great Britain (Ridgill and Fox 1990). The national totals of some species also vary with changing annual breeding success. For example, unsuitable breeding conditions decrease recruitment, thereby decreasing the total winter population estimate. On a national scale the aim is, therefore, not necessarily to maintain a constant number of birds within the protected site network. Rather, the network should act as a buffer against changes at national level, such that if the national total declines, the proportion of birds in the network should increase (or at the very least remain constant). In this respect, for those species where the national totals have decreased, a positive correlation between the proportion of the national total in the PS network and the size of the national total would suggest that the protected network was losing individuals disproportionately. However, amongst the 21 species examined none of the species decreased between 1975/76-1998/99. Considering the period between 1990/91-1998/99 both dark-bellied brent goose and mallard actually decreased, however, correlations between the proportion of the national total within the PS and the size of the national total were significantly negative. Indeed, for those species whose national totals have either remained stable or declined, a negative correlation between the proportion of the national total supported by the PS network and the overall numbers nationally is highly encouraging as it indicates that the PS are retaining numbers despite an overall decline nationally. In this instance, changes are preferentially occurring on sites outside the PS network. This is especially relevant for mallard, as recent trend analyses have suggested that numbers in the UK have decreased by 20-30% over the period 1994/95-1998/99 (Kershaw and Cranswick 2003).

Alternatively, for those species whose national totals have increased over time, a negative correlation indicates that the proportion of birds within the currently designated PS network has decreased over time. In this respect, gadwall showed the most significant negative correlation and has also experienced the most dramatic increase in national numbers since 1975/76. Although such a relationship could be indicative of a shift in preference away from the PS network and a reflection of the failure of protected areas to provide suitable protection measures for this species, it is also possible that the protected areas have simply become saturated. Specifically, once the protected areas reach capacity numbers will, of necessity, increase disproportionately elsewhere. Over time, this will result in a negative correlation between the national total and the proportion of this total within the PS. In this respect, a recent analysis of the status of RSPB reserves reports an annually increasing number of avocets away from protected areas following the stabilisation of numbers on reserves (RSPB 2002).

Where the proportion of the national total within the PS network remained fairly constant over time, despite substantial fluctuations in numbers nationally (e.g. Bewick's swan, European white-fronted goose and shelduck), the likelihood is that demographic processes operating at a local level are more important than regional, wider countryside factors in structuring PS totals (also discussed in Chapter 9). Specifically, by applying the appropriate management efforts, for example simply by increasing the area of suitable habitat within an individual protected area, a greater proportion of the annual national total of a particular species could be attracted to a single site in a given year. In this respect, in Great Britain efforts have been made with much success over the past few years to encourage overwintering geese populations away from arable land by improving the quality of alternative grassland sites (Vickery et al. 1997). Conversely, where there were large fluctuations in the proportion of individuals in the PS network (e.g. whooper swan, dark-bellied brent goose, smew, red-breasted merganser and goosander) the likelihood is that regional, wider countryside factors are more important. Specifically, in any one year, the distribution of individuals across wintering sites is determined by large-scale environmental processes such as the breeding conditions and weather conditions along migration routes rather than local site characteristics (Gill et al. 2001, Goss-Custard et al. 2002, Webster et al. 2002). For these species, it is unlikely that the management of sites in isolation will be especially effective. Rather, to be successful, management actions should be focussed at the national or network level (see Chapter 9).

3.1.4 Conclusions

The results of this analysis, in contrast to the majority of published studies of the performance of protected areas, provide a largely encouraging evaluation of the current status of SPAs and Ramsar Sites in Great Britain. Indeed, for all 21 species, the LS contain significantly more individuals compared with random sets. By contrast, considering the PS network, for each of the 21 species analysed, at least one randomly generated combination of wetland sites supported greater numbers of birds than the existing PS network. In addition, for certain of the 21 species, these analyses demonstrate the need to re-evaluate the existing SPA/Ramsar Site network, to ensure that, should the recent declines recorded nationally continue, numbers within these PS, and more importantly in the LS suite, will remain at least stable despite decreasing numbers in the wider countryside.

3.2 Protected versus non-protected areas: Rates and Means

3.2.1 Introduction

It has been strongly argued that, because of inadequacies inherent in the majority of existing biological data sets, current protected area networks cannot be realistically evaluated until supplementary distribution and abundance data are collected (Prendergast et al. 1999). However, given that the organisation and execution of data collection is time consuming and resource intensive, there is a risk that by the time a database can be considered sufficiently comprehensive it is likely that allocated budgets will have been exhausted and the species under consideration have declined below a recoverable level. In general, therefore, it will not be sensible or indeed feasible to delay conservation programmes until data collection is complete (Pressey and Cowling 2001, Williams et al. 2002). Indeed, the budget allocated/available (or not) for conservation more often than not relates to politics rather than biological urgency, election times being especially effective for swaying decisions in favour of conservation schemes (Marren 2002; see also Chapter 10). In consequence, conservation planners are required to make the best use of those data that are currently available (Margules and Austin 1994, Pressey and Cowling 2001, Margules et al. 2002, Williams et al. 2002).

Nonetheless, inappropriate use of existing data could have extensive consequences for biodiversity conservation (Freitag and van Jaarsveld 1998). In many cases this conclusion has been arrived at through trial and error (Wilhere 2002), leading to the realisation that many existing protected areas are of limited conservation value (Pressey 1994, Pressey and Tully 1994, Freitag et al. 1998, Cabeza and Moilanen 2001, Pressey and Cowling 2001). Despite increasingly urgent calls for the development and application of systematic priority-site selection approaches (see for example Rodrigues and Gaston 2001, Gaston et al. 2002, Thiollay 2002), and repeated calls for comprehensive evaluation systems (see Hocklings et al. 2000 and the references therein), critical evaluations of the performance of protected area networks remain surprisingly rare (although see Table 1.4). This evaluation is, however, vital, not only to determine the status of existing protected areas, but also as part of an ongoing feedback system, progressively to improve the quality and viability of protected areas for biodiversity preservation and to inform active management (Cabeza and Moilanen 2001, Wilhere 2002).

The JNCC numerical selection criteria for SPA and Ramsar Sites in Great Britain relate not only to the preferred characteristics of potential protected areas (described in detail in Chapter 2), but also to specific management requirements and the need for appropriate and regular monitoring procedures (Chapter 3.1). Furthermore, every three years, each Member State of the European Union is required to submit a progress report, detailing information pertaining to the status of both individual sites and the network as a whole (Article 12). Following submission of this report, any country failing to show satisfactory progress, or where the quality of SPAs has declined markedly, faces the prospect of legal action by the European Court of Justice, as has been the experience of Sweden (1995), France (1999), Greece, Austria and Portugal (2000), and Ireland (2002).

As a result of the numerical site selection guidelines for the identification of SPAs and Ramsar Sites (Protected Sites (PS)), and in view of the legal requirement for appropriate site management at the species level, the expectation is that the PS network and in particular those SPAs classified as LS for individual species, will have performed better in sustaining waterbird populations than other non-PS wetlands. To test this assumption, we compared the mean numbers of individuals and the rates of population size change for 13 species of migratory waterbird on non-PS, PS and LS wetlands using data from 1990/91-1998/99.

3.2.2 Methods

Data

Data for the nine year period, 1990/91 (earliest point where there were sufficient numbers of PS and LS officially classified for all 13 species) to 1998/99 (latest available computerised data as of September 2001) inclusive, were included for Bewick's swan, European white-fronted goose, dark-bellied brent goose, shelduck, wigeon, gadwall, teal, mallard, pintail, shoveler, pochard, tufted duck and goosander. These 13 species are those for which WeBS data are the primary source of information for SPA/Ramsar Site classification/designation. The summed data for the 13 species over this same period were also included.

National totals

For each of the 13 species individually, following the procedure established by WeBS (Pollitt et al., 2000), the national total (numbers of birds recorded nationally) for a given month was take to be the sum of the individual monthly counts across the 1962 selected WeBS sites in Great Britain. The peak of these monthly counts in each year was taken to represent the national total for each species in each year (1990/91-1998/99).

Individual site totals

For the 13 species separately, counts from each WeBS count unit were summed to provide estimates for each site. The maximum number of individuals recorded on each of these sites between September and March was then determined for each year 1990/91-1998/99. In addition, for each species, totals for all PS and those sites listed as nationally important (sites supporting ≥ 1% national population of a species or ≥ 2,000 birds) were summed to provide annual counts for the LS suite.

The PS network is not static and new sites are classified/designated in each year (Fig. 2.1). These analyses (unless otherwise stated) were, therefore, carried out using only those sites classified as prior to and including 1990/91 (*N* = 138). The annual totals for each species, therefore, represent the totals for this set of sites from 1990/91-1998/99. This means that SPAs or Ramsar sites identified after this date will not be included. It should be borne in mind that the PS network in Great Britain is constantly evolving and will now include additional sites, but given that data were not available these have not been included here. This means that the numbers of PS or LS included will not equate to current totals published by the JNCC in the recent SPA review (Stroud et al. 2001; Table 3.2.1).

Table 3.2.1: The total numbers of WeBS sites, numbers of Protected Sites (PS) and numbers of Listed Sites (LS) included in the analysis for all 13 species summed and for each species individually. These sites are those for which at least one individual was recorded between 1990/91 and 1998/99. x No LS.

Species	National	PS	LS
All 13 species together	1962	136	x
Bewick's swan	204	19	11
European white-fronted goose	152	17	4
Dark-bellied brent goose	140	17	5
Shelduck	385	28	7
Wigeon	330	28	12
Gadwall	426	24	5
Teal	643	27	10
Mallard	704	30	6
Pintail	378	29	11
Shoveler	474	28	7
Pochard	619	30	5
Tufted duck	654	29	3
Goosander	546	24	x

Analyses

For each WeBS site, in accordance with WeBS methodology, the mean numbers of individuals (henceforth referred to as 'site means'), both for each of the species individually and the summed total across all 13, was taken as the five-year mean from the individual site counts between 1990/91-1994/95 and also between 1994/95-1998/99. However, because the results were largely identical, only those from 1994/95-1998/99 are reported here. The data for these analyses appeared to violate normality assumptions, therefore, comparisons between the PS and non-PS and PS and LS wetlands were made using the non-parametric Mann-Whitney U test.

For each species individually, the rates of population change for each site (henceforth termed simply 'rates of change') were calculated for the period 1990/91-1998/99. For each of the 13 species, a set of scatter plots (one for each site) was generated, with the local site population plotted for each year. For each wetland, the rate of change was taken as the regression slope of the individual scatter plots. Various regression models (Minitab ™ version 13) were tested, however, in each case linear regression resulted in a significantly lower residual sum of squares term and was therefore considered the most suitable. For each wetland, the rates of change were calculated using both the raw numbers of birds supported by each wetland site (absolute rates) and the $\log_{10}$ local site numbers (relative rates).

Finally, the total numbers of birds gained/lost for each of the species individually (dependent on the direction of the national trend; see Chapter 3.1) was taken as the summed total of the absolute rates of change per year for each wetland site where a species was recorded.

3.2.3 Results

Site means

All 13 species

Considering the collective total for all 13 waterbird species, there was a significant difference in the mean numbers of individuals supported by PS and non-PS wetlands (U = 160085.5; N = 1962; $P < 0.0001$).

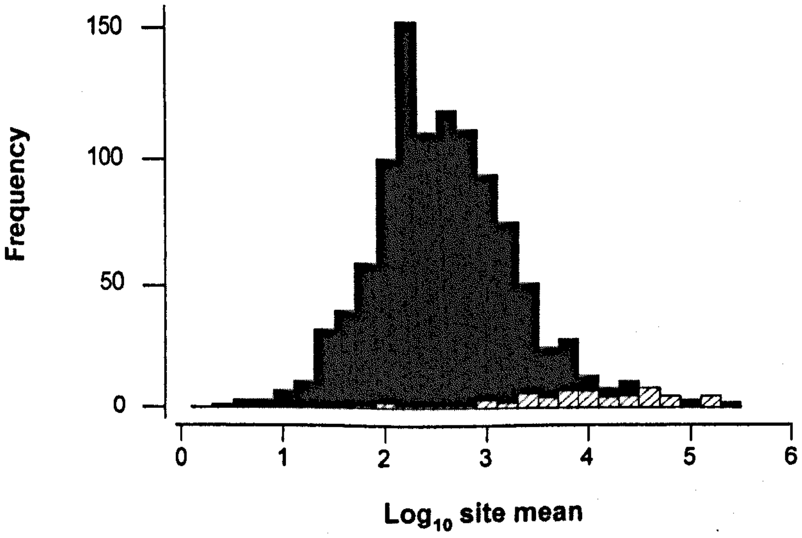

Figure 3.2.1: Histogram showing the distribution of $\log_{10}$ mean numbers of individuals across all non-PS (dark bars) and PS wetlands (hatched bars) for the collective total numbers of all 13 species. NB. The combinations of filled and hatched bars are independent with respect to the y-axis.

Greater numbers of birds were recorded on the PS network, with a mean (±SD) of 6,087.7 (±11,873.4) birds compared with 327.8 (±918.9) birds across the non-PS wetlands (Fig. 3.2.1). However, there was a greater standard deviation from this mean number for the PS, resulting from a greater range of site means for the PS wetlands (1.4 to 100,829.4 birds) compared with the non-PS wetlands (0.2 to 22,343.8 birds).

Individual species

Taking each of the 13 waterbird species separately, for seven species the top site for mean numbers of individuals was included in the PS network. Further, for six of these this top site was also listed as nationally important (LS). Nonetheless, for eight species, less than half the top ten sites ranked in descending order of the numbers of individuals were included in the PS network. Additionally, for nine species less than half the top ten sites was part of the LS suite.

Table 3.2.2: Comparisons (Mann-Whitney U test) of the mean numbers of individuals supported by a site and the rates of population change (absolute and relative) across sites for each of the 13 species of waterbird individually between: a) PS and non-PS wetlands and b) PS and LS wetlands. *$P < 0.05$, **$P < 0.01$, *** $P < 0.001$, **** $P < 0.0001$. ^{x}No LS. (Numbers of sites included in the analysis for each species are listed in Table 3.2.1).

Species	**Site means**		**Rates of change**			
			Absolute		**Relative**	
	a	**b**	**a**	**b**	**a**	**B**
All species	3822****		108221		109982	
Bewick's swan	22619**	28*	1096**	40	1257*	36
European white fronted goose	7567*	15**	1029	15	949	14
Dark-bellied brent goose	4702	67***	526***	2***	659**	28
Shelduck	24006**	15*	4907	61	4729	67
Wigeon	14922**	10**	3025**	91	2767**	72
Gadwall	30828***	10*	4049	25	4695	45
Teal	43638****	10**	7484	72	8550	44
Mallard	4698****	28*	6443***	36	9076	64
Pintail	25919**	18	4786	88	4875	88
Shoveler	36956***	6*	6407	69	6283	62
Pochard	44999***	21*	6271**	29	7326	24*
Tufted duck	45419**	21*	7355	26	8242	31
Goosander	35129		5628		5686	

For ten of the 13 species analysed there were significant differences in the site means between PS and non-PS wetlands. In each case, greater numbers of birds were recorded on the PS wetlands (Table 3.2.2). For example, for mallard the overall mean (±SD) across the individual site means was 288.5 (±447.0) birds for non-PS wetlands and 1,258.8 (±996.9) birds for the PS (Fig. 3.2.2a). In contrast, for European white fronted goose, dark-bellied brent goose (Fig. 3.2.2b) and goosander there were no significant differences in site means between the PS and non-PS wetlands (Table 3.2.2).

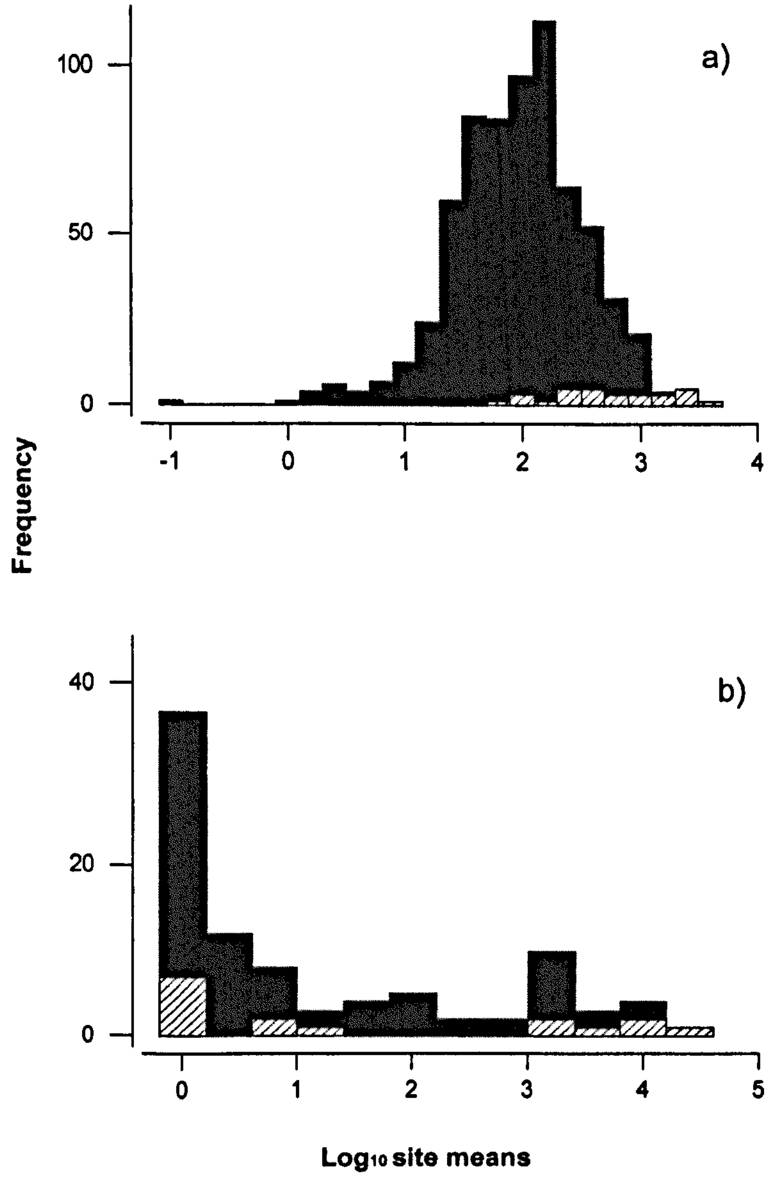

Figure 3.2.2: Histogram showing the distribution of $\log_{10}$ mean numbers of individuals across all non-PS (dark bars) and PS wetlands (hatched bars) for a) mallard and b) dark-bellied brent goose.

Considering comparisons between the PS and the LS for each species, there were significant differences in the mean numbers of individuals supported for nine of the 12 species (no LS officially classified for goosander), although for eight of these this was borderline (Table 3.2.2). However, for each, greater numbers of birds were recorded on the LS. For example, for wigeon the overall mean (±SD) numbers of birds for the PS network was 7,880.7 (±9,861.1) birds compared with 13,783.2 (±9,818.2) birds for the LS. Additionally, for shelduck, mallard and pintail, despite the lack of significant differences, the overall mean number of birds was greater for the LS.

Rates of change

All 13 species

Taking the total numbers of birds across all 13 species, there was no significant difference between the rates of change (absolute or relative) for PS and non-PS wetlands (Fig. 3.2.3; Table 3.2.2). In general, however, the absolute rates of change were faster for PS (mean (±SD) 20.5 (±0.8) birds per year) compared with the non-PS wetlands (0.12 (±0.02) birds per year; Fig. 3.2.3). The overall increase in numbers for all 13 species in each year between 1990/91-1998/99 (850.5 birds per year; b = 19118.4 (±9591.35) was distributed approximately equally between the PS and non-PS wetlands (Table 3.2.3; Fig. 3.2.3). Nonetheless, the total numbers of birds gained across all wetlands (both PS and non-PS) amounted to only 0.64% of the 1990 total (Table 3.2.3).

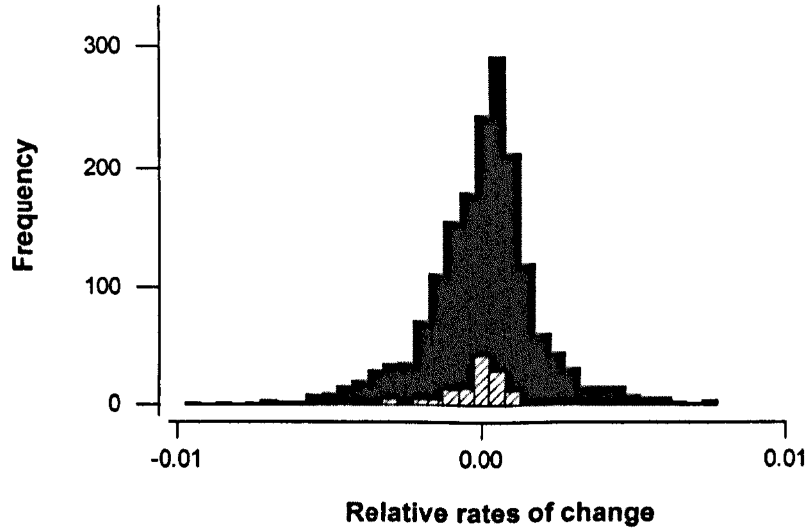

Figure 3.2.3: Histogram showing the distribution of the rates of population size change across all non-PS (dark bars) and PS wetlands (hatched bars) for all WeBS species together.

Table 3.2.3: The total number of birds gained per year (absolute rates of change); the % of this total from the non-PS and PS; the % of the total gained from the PS from the LS wetlands; the 1990/91 total number of individuals; the % of this total from the non-PS and PS; and the % of the total gained from the PS from the LS wetlands for each of the species individually and for all 13 species together where the national trend has increased.

Species	Total gains per year	Non-PS (%)	PS (%)	LS (% from PS gain)	1990 total	% 1990 total gained from		
						Non-PS	PS	LS (% PS gain)
All species	850.48	49.06	50.94		1193423	0.31	0.33	
European white-fronted goose	65.14	48.22	51.78	33.90	5882	4.81	5.16	2.65
Wigeon	16381.38	85.91	14.09	47.06	355543	35.62	5.84	5.19
Gadwall	1216.23	88.71	10.51	52.65	11763	82.55	10.51	4.82
Teal	5214.38	85.76	14.24	45.74	188088	21.40	3.55	3.00
Pintail	1472.44	58.48	41.52	47.23	36114	21.46	15.24	13.63
Shoveler	720.97	87.89	12.11	41.99	15362	37.12	5.12	3.70
Pochard	2100.06	61.32	38.68	46.32	51710	22.41	14.14	12.20
Tufted duck	2797.77	87.34	12.66	36.64	63711	34.52	5.00	2.89
Goosander	379.08	92.01	7.99	*	4562	68.81	4.53	*

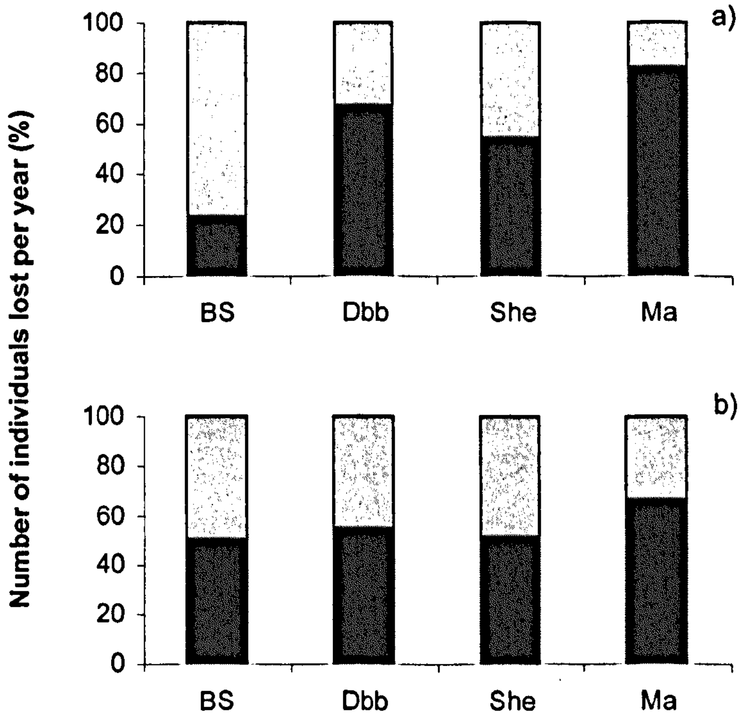

Figure 3.2.4: The proportion of the total number of individuals lost per year for: a) non-PS (black bars) and PS wetlands (grey bars); and for b) PS (black bars) and LS (grey bars) wetlands for: all 13 species collectively (All), Bewick's swan (Bs), European white-fronted goose (Ewf), Dark-bellied brent goose (Dbb), Shelduck (She), Wigeon (Wi), Teal (Te), Mallard (Ma) and Pintail (Pi).

Individual species

The absolute numbers of individuals lost per year from the PS for those species for which the national total decreased over the period 1990/91-1998/99 (See Chapter 3.1) ranged from 17.9% of the national total for mallard to 76.6% for Bewick's swan. Nonetheless, for each of the four species, less than half these individuals were lost from the LS wetlands (Table 3.2.4; Fig. 3.2.4).

The total numbers of individuals for each species lost from the PS between 1990/91-1998/99 ranged from 6.9% of the overall numbers lost nationally for mallard, to 36.5% for Bewick's swan (Table 3.2.4; Fig. 3.2.4a). Of these PS losses, the percentage lost from the LS ranged from 9.3% for mallard to 48.9% for Bewick's swan (Table 3.2.4). In addition, for all four species, less than half the total numbers of individuals lost from the PS network between 1990/91-1998/99 were from LS wetlands. For example, for Bewick's swan, 6681.02 birds were lost across all PS wetlands from 1990/91-1998/99 (742.34 birds per year), of which 3265.79 individuals were from the LS wetlands.

Table 3.2.4: The total number of birds lost per year (absolute rates of change); the % of this total from the non-PS and PS; the % of the total lost from the PS from the LS wetlands; the 1990/91 total number of individuals; the % of this total from the non-PS and PS; and the % of the total lost from the PS from the LS wetlands for each of the species individually where the national trend has decreased.

Species	Total loss per year	Non-PS (%)	PS (%)	LS (% from PS loss)	1990 total	% 1990 total lost from		
						Non-PS	PS	LS % PS loss)
Bewick's swan	495.54	23.42	76.58	48.88	9359	11.16	36.49	48.88
Dark-bellied brent goose	5884.67	67.31	32.69	43.31	154145	23.13	11.23	9.31
Shelduck	2767.99	54.73	45.27	48.48	100406	13.58	11.23	10.57
Mallard	8422.77	82.08	17.92	34.11	196778	31.62	6.9	3.57

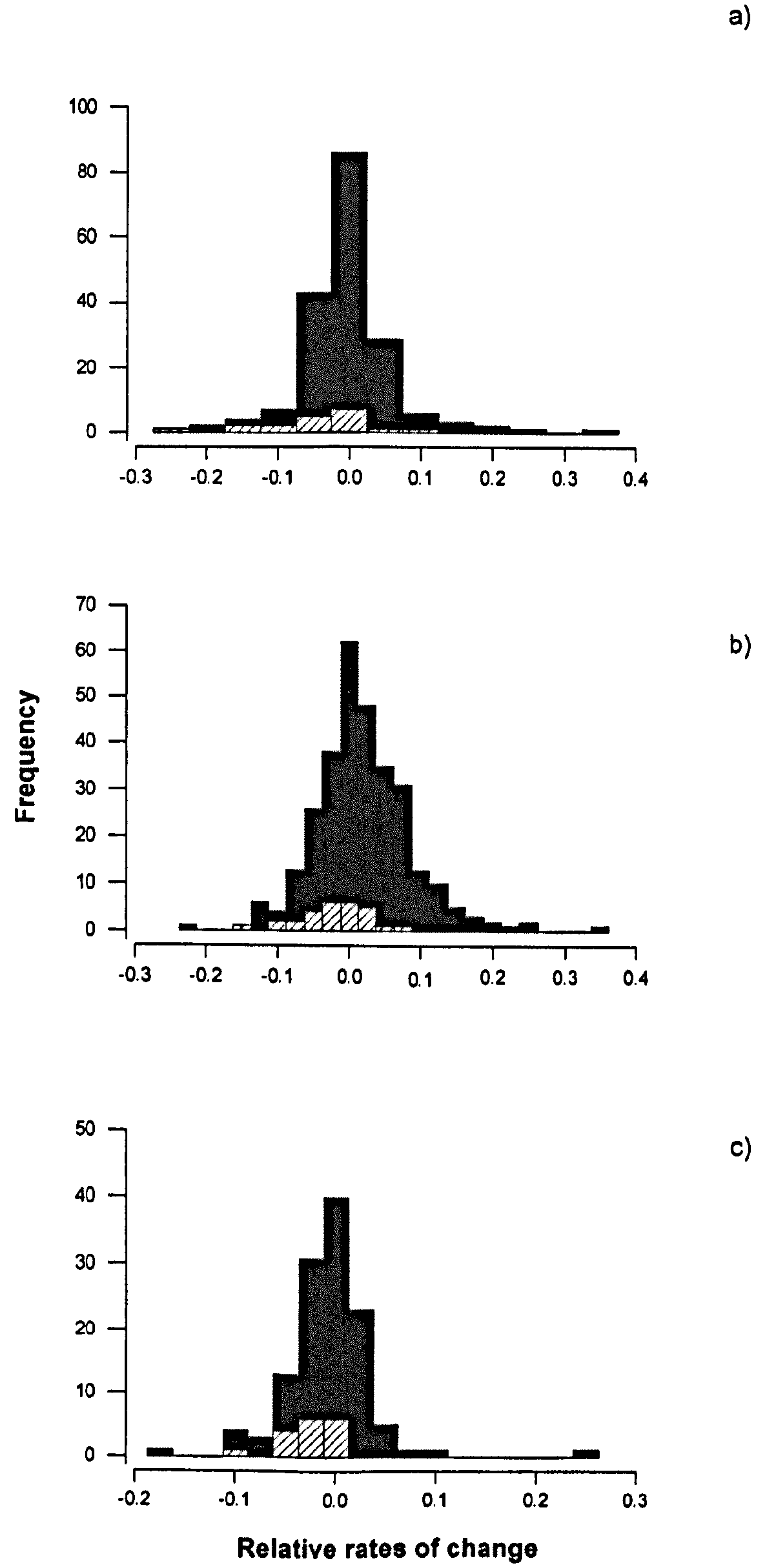

Figure 3.2.5: Histograms showing: a) the relative rates of change across all non-PS (dark bars) and PS wetlands (hatched bars) for wigeon; b) dark-bellied brent goose; and c) the distribution of the rates of change across all PS (dark bars) and LS (hatched bars) for dark-bellied brent goose.

Similarly, for mallard, 20618.28 birds were lost over this nine year period (2290.92 birds per year) of which 7033.5 individuals were lost from the LS. The national total increased over the period 1990/91-1998/99 for nine species (Chapter 3.1). The total numbers of individuals gained per year by the PS network ranged from 8.0% of the total numbers gained nationally for goosander, to 51.9% for European white fronted goose (Table 3.2.3). Further, for eight of the nine species, less than half these annual increases in numbers were from PS wetlands (Fig. 3.2.4a).

The proportion of the annual gain in individuals on the PS network from LS wetlands was in all cases > 30% and varied from 31.4% for gadwall to 54.7% for teal (Fig. 3.2.4b; Table 3.2.3). The total numbers of individuals of each species gained by the PS between 1990/91-1998/99 ranged from 3.6% of the 1990/91 population for teal, to 15.2% for pintail (Table 3.2.3). In addition, the percentage of this increase in numbers for the PS network from the LS for a particular species was always <15%, and ranged from 2.7% for European white-fronted goose to 13.6% for pintail (Table 3.2.3). For example, for Bewick's swan 95.6% of sites where the total number of individuals decreased were LS, which equates to a loss of 35.0% (3265.8 birds) of the 1990/91 total.

Overall, the site with the fastest positive absolute rate of change (i.e. gaining individuals at the fastest rate) was included in the PS network for only four species (European white-fronted goose, shelduck, pintail and goosander). Additionally, for all species no more than four of the top ten sites, ranked in descending order of their absolute rates of change, were included in the PS network, and for dark-bellied brent goose none of the ten most rapidly increasing sites were PS wetlands. However, considering the relative rates of change, the top site was a PS (sites ranked in descending order of their relative rates of change) for European white-fronted goose alone and for ten for the species none of the top ten sites were PS.

There were significant differences in the absolute rates of change between PS and non-PS wetlands for only five of the 13 species (Bewick's swan, dark-bellied brent goose, wigeon, mallard and shoveler). Furthermore, for each the absolute rates of change for non-PS wetlands were faster (more additional birds in each year) than for the PS (i.e. the non-PS wetlands have gained more individuals in each year than the PS wetlands; Table 3.2.2; Table 3.2.3). Nonetheless, considering the relative rates of change (i.e. where the effect of population size is removed), these rates were significant for only three of the five species (Bewick's swan (Fig. 3.2.5a), dark-bellied brent goose, and wigeon (Fig. 3.2.5b)). In each case, the relative rates of change were slower for PS as opposed to non-PS wetlands.

Comparisons between the absolute rates of change for PS and LS wetlands were significant for dark-bellied brent goose alone (Table 3.2.2). The rates of change were significantly slower for those PS listed as nationally important (LS) (-318.8 (±94.5) birds per year as opposed to -27.4 (±74.5) birds per year for the PS wetlands). Further, losses from the LS accounted for 82.9% of the total lost for the PS network. Nonetheless, there were no significant differences between PS and LS wetlands in terms of their relative rates of change (Table 3.2.2).

3.2.4 Discussion

Given the inevitable conflicting demands on wetlands from economic, social and conservation perspectives (Chapter 1), one could expect that analyses of the performance of the SPA/Ramsar Site network in Great Britain would be particularly disheartening. Further, of the few published examples of existing protected area evaluations, the majority conclude that target species representation or network coverage in general is inadequate (Table 1.4). For example, Rebelo and Siegfried (1992) found that existing reserves set aside for floral diversity in the Cape Floristic Region of South-Africa contained no more species than predicted by a null model. Similarly, Khan et al. (1997) concluded that 83% of threatened plant species in New Caledonia were not found within existing protected areas. Nevertheless, in contrast to this reported lack of effectiveness for other protected area networks, without exception the SPA/Ramsar Site network supported significantly greater mean numbers of individuals compared with non-PS wetlands. This is highly encouraging, particularly given that not all species present on a site will have been included in the initial selection process due to not meeting the threshold values stated in the SPA/Ramsar Site selection criteria (see also Chapters 2 and 3.1). Nonetheless, it should be borne in mind that there are several alternative explanations for these significant differences. First, the greater mean numbers of birds supported by the PS network may simply reflect the superiority of these sites in the first place. Fundamentally, because the numerical approach to SPA/Ramsar Site selection deliberately aims to identify those sites where the numbers of target species are greatest, such significant differences are inevitable and, therefore, do not necessarily reflect the success of management.

Second, the PS may simply be the largest wetland areas and are able, therefore, to support greater numbers of individuals. This is both a simple function of area, larger areas are able to support more individuals, and also because larger areas are more likely to incorporate a greater habitat and resource diversity than smaller areas (Gaston et al. 2002). This increased habitat diversity will, therefore, enable a wider range of species and greater overall numbers of individuals to winter at a particular site.

Sites listed as nationally important for a species (LS) also supported significantly greater numbers of birds when compared to the remaining PS for that same species. Naturally, this could be for the same reasons detailed above. Nonetheless, as for the PS/non-PS comparisons, an important factor of note is that these sites have remained superior sites since their classification/designation as SPAs/Ramsar Sites. Although, this is not necessarily indicative of the success of a particular management strategy, at the very least it suggests that any activities have not had an adverse or deleterious effect on wintering waterbird numbers.

In contrast to the greater mean numbers of birds supported by PS and LS wetlands, considering each of the 13 species individually, a large percentage of the rates of change for the PS/LS were negative (i.e. losing individuals over time) compared with the non-PS/non-LS. However, the contribution made by these decreasing PS/LS wetlands to the overall numbers of birds lost for a particular species varied considerably. This contribution is especially relevant for those species where the national total has decreased (Bewick's swan, dark-bellied brent goose, shelduck and mallard). Indeed, for such species, a substantially greater proportion of the overall loss of individuals from the PS wetlands as opposed to the non-PS wetlands indicates that the decline in numbers is occurring disproportionately from the PS network. Indeed, for Bewick's swan 68.4% of the PS have decreased in numbers, which contributed to more than 75% of the annual losses between 1990/91-1998/99. Furthermore, the projected losses over the nine year period amounted to greater than 30% of the 1990/91 total numbers of birds. This is of particular conservation concern as it indicates that the network of protected areas is not having the desired effect. Perhaps of more immediate conservation concern, is that numbers of birds for 72.7% of the LS (sites listed as nationally important) have also decreased. Given that these sites are those recognised as nationally important and are, therefore, formally required to adopt measures for individual species, particularly those listed on Annex I of the Directive, these results are far from encouraging.

In contrast to the large proportion of the annual loss of individuals from the PS wetlands for Bewick's swan, for dark-bellied brent goose and mallard a greater proportion of these losses were from the non-PS wetlands. Specifically, for mallard although 80% of the PS decreased, this amounted to only 17.9% of the overall loss of individuals. Similarly, for dark-bellied brent goose, 82.3% of the PS lost individuals over time, however, these sites accounted for less then 35% of the overall numbers lost in each year. Given that the protection of key habitat areas is one means by which the problem of declining species can be addressed this is a positive sign and suggests that PS are in fact providing suitable refugia for such species. Not only does this mean that managers could potentially influence an increasing proportion of the national population at the local site level, but it also suggests that management of these sites has been effective (see Chapter 3.1).

For dark-bellied brent goose, of concern is that the numbers of individuals on all the LS for this species decreased between 1990/91-1998/99. Further, despite the smaller number of sites in the LS suite compared with the PS network, this decrease accounts for almost half the total numbers lost from the PS network each year. Given that these sites are considered of national importance, and for which management efforts are required specific to individual species, the expectation is that these sites should be refugia for a declining species. The implication, therefore, is that numbers should not decline and ideally should remain stable (Chapter 8). However, for these wetlands it would appear that this is not the case. This would suggest that these sites are not the most suitable sites for this species, either because of historical distributions and site fidelity in order to prevent declines to unsustainable levels (Chapter 2), or inappropriate/inadequate management activities. It will, therefore, be necessary to re-evaluate these sites to ensure numbers do not decline further.

In contrast to the negative trend in the numbers of birds wintering on wetlands for four species, the national total increased for nine species. Nonetheless, there were no significant differences in the relative rates of change between PS and non-PS wetlands for eight of these species. This suggests that, for these species, numbers of birds have increased across all suitable sites rather than just on the PS network. Indeed, for each species approximately half the PS and the non-PS wetlands increased in numbers. By contrast, for wigeon, the relative rates of change for PS wetlands were slower than for non-PS wetlands. Further, 85.9% of the individuals gained per year were from the non-PS wetlands rather than the PS network. It should be borne in mind, however, that there are several alternative explanations for the greater rates of increase on the non-PS wetlands as opposed to the PS network.

First, and perhaps of most concern, is that the LS are not actually being managed for the benefit of wigeon. Indeed, because individual SPAs are often listed as nationally important for more than one species, conflicts between species in terms of their management requirements are likely (discussed in detail in Chapter 7). Therefore, with the exception of the largest wetland sites where multiple management options can be addressed simultaneously, it may be that these sites are being actively managed for species whose requirements are directly at odds with those of wigeon. For example, management of a site for a species such as goosander, which requires deeper, slow flowing water conflict directly with wigeon habitat requirement for standing shallow waterbodies (Brunn et al. 1987; Mullarney et al. 1999).

Second, it may be that, following sustained population increases nationally, the PS and LS have reached capacity and, following Brown's concept of the buffer effect (Brown 1969), additional individuals have been forced to relocate to alternative areas (discussed in detail in Chapter 8). The expectation for the buffer effect is that the original sites then remain stable as any changes are effectively buffered by these alternative areas (Moser 1988, Halama and Duesser 1994, Ferrer and Donazar 1996, Gill et al. 2001, Vickery 2001). However, in this case, more than half the PS actually lost individuals over the nine year period 1990/91-1998/99. One possible explanation is that individuals do not move alone, and therefore, relocate to new areas in groups. This effect would then be reinforced through conspecific attraction, where individuals arrive at an area because they were attracted by the presence of conspecifics (Brown et al. 2000). Thus, more individuals than expected would vacate the original site (Chapter 8).

3.2.5 Conclusions

The few previous studies that evaluate the performance of existing networks have concluded that coverage/representation is inadequate. Furthermore, given the lack of explicit site selection criteria included within the Birds Directive and the inevitable constraints on conservation efforts from competing land use, the success of the PS network for these 13 species is a considerable achievement (see also Chapters 2 and 3.1). Nonetheless, although for the majority of species these results are extremely encouraging, of concern are those species for which declines are occurring disproportionately for the PS/LS compared with the wider countryside. In particular, for Bewick's swan, dark-bellied brent goose and mallard, there is a real need for a re-evaluation of the PS and LS network, to determine the reasons for these declines and to act accordingly. However, for all SPA/Ramsar Site target species, including the 13 analysed here, it will be of particular importance to continue accurately to monitor individual populations both on and off SPAs and Ramsar Sites, so that any undesirable changes can be detected at the earliest possible stage. In this way, more time will be available for the appropriate remedial action to be employed. This negative trend can be seen as an early warning that it is likely that these species will begin to decline significantly at a later date.

CHAPTER 4: NETWORK EFFECTIVENESS

"We cannot solve the problems we have created with the same thinking that created them."
Albert Einstein 1879-1955

Although Chapters 2 and 3 demonstrate the largely excellent performance of SPAs and Ramsar Sites in Great Britain for wintering waterbirds, they also reveal a bias towards aggregating species such as dark-bellied brent goose and European white-fronted goose. Given that all species listed in Annex I or II of the Birds Directive are necessarily priorities for conservation efforts this should be seen as a failing of the current network adequately to satisfy the objectives of the Directive. Following from the recommendations of these two chapters (i.e. the adoption of species specific representation targets and the consideration of networks of sites as opposed to each priority area in isolation), this chapter looks at the effectiveness and efficiency of the PS network using linear programming techniques to assess current, future and potential species representation. Chapter 5 then explores the priority site selection process itself and uses linear programming algorithms to determine the optimal data requirements for SPA/Ramsar Site selection.

The performance of procedures for selecting conservation areas

4.1 Introduction

The establishment of protected areas for conservation is a requirement of parties to a number of international conventions, including the Convention on Biological Diversity, the Berne Convention on the Conservation of European Wildlife and Natural Habitats, and the Ramsar Convention on Wetlands of International Importance. However, time and budget constraints, incomplete data, issues of ownership, conflicting land use, historical designations, and limited public and institutional support often complicate their practical implementation (Bishop et al. 1995, Clarke 2000). A necessary requirement of networks created following such international legislation is the long-term *in situ* preservation of "biological diversity", or some component thereof, specifically in response to increasing rates of species extinction and habitat degradation (Pressey 1994, Stockland 1997, Hopkinson et al. 2000a, Rodrigues et al. 2000b). Nonetheless, the success of a conservation network, in relation to the persistence of the species it aims to protect, is to an extent determined by the methods used to identify the constituent sites. Inevitably, the particular species and/or habitat management tools employed once a site has been selected are also important. The potential of these management tools is essentially limited by the capability of the site-selection methods to distinguish the most appropriate sites to begin with. Such methods are required to determine reliably those sites where, in combination and with appropriate management, the target species will persist in viable numbers into the future.

In recent years, much attention has been directed to the ways in which the most important areas for inclusion in reserve networks can be identified, and the most effective ways in which existing networks can be expanded (Rebelo and Siegfried 1992, Pressey et al. 1993, Sætersdal et al. 1993, Williams et al. 1996, Stockland 1997, Rodrigues et al. 1999, Margules and Pressey 2000, Fairbanks et al. 2001). In contrast, rather little attention has been paid to the present and likely future performance of the approaches that have actually been employed. In addition, where this has been done, the emphasis has typically been on the persistence of species within a network rather than on the maintenance of population sizes, and on hypothetical networks selected using commonly applied methods rather than those networks that have actually been selected (see Margules et al. 1994, Kunin 1997, Pressey et al. 1997, Nicholls 1998, Rodrigues et al. 1999, Rodrigues et al. 2000b, Chown et al. 2001, Verboom et al. 2001).

Through the application of numerical thresholds for population size for the selection of nationally and internationally important wetland sites for wintering waterbirds, Chapters 2 and 3 demonstrate a bias towards those species that form dense aggregations. By contrast, those species which exhibit a more dispersed distribution across their wintering sites tend to be under-represented within the current PS network. To redress the balance, therefore, it is necessary to develop alternative means by which priority sites are selected. In this respect, this Chapter examines the performance of the overall network of SPAs and Ramsar Sites (Protected Sites (PS) network) in maintaining populations of waterfowl in Great Britain over a 23 year period (1976/77-1998/99), relative to the performance of hypothetical alternative networks selected using other approaches, and suggests ways in which the PS network may be improved through the application of various alternative site-selection strategies.

4.2 Conservation planning and priority-site selection

Priority-site selection algorithms are sets of rules systematically designed to achieve particular goals with maximum efficiency (Pressey et al. 1993, Csuti et al. 1997, Williams and ReVelle 1997, Pressey 1999, Margules and Pressey 2000, Rodrigues et al. 2000a). Over the past decade, the evolution of mathematical approaches has progressed from simple scoring, where sites were ranked relative to features such as total abundance and rarity, to iterative heuristic methods, and finally to linear integer programming. Although each of these methods differ in the objectives they emphasise and the algorithms used, all select sites in an explicit, objective, repeatable, and efficient manner (Bedward et al. 1992, Nicholls and Margules 1993, Pressey et al. 1996). That is, to select a reserve system that accomplishes the most protection for the least cost (or area). Nonetheless, while scoring and heuristic methods follow procedures designed to achieve efficiency, they cannot guarantee an optimal solution, nor can they measure how far from optimality they are (Pressey and Nicholls 1989a, Vane-Wright et al. 1991, Underhill 1994, Camm et al. 1996, Church et al. 1996, Williams and ReVelle 1997, Cabeza and Moilanen 2001). Furthermore, the only way to determine the degree to which a solution derived using heuristic methods is sub-optimal is to compare it with that obtained using exact algorithms. If this is feasible there is little value in the original heuristic approach.

A recognised alternative to heuristics is the use of exact algorithms. This is the use of integer programming software to solve the explicit constraints and objectives of a problem (Williams and ReVelle 1997). Indeed, such equations are being used more regularly as scientists begin to accept their value in terms of attaining selection accuracy and optimality (Cocks and Baird 1989, Sætersdal et al. 1993, Church et al. 1996, Willis et al. 1996, Rodrigues et al. 2000a, 2000b, 2000c).

There are essentially two formally established approaches to linear programming: the minimum area problem (Pressey et al. 1997) and the maximal coverage problem (Arthur et al. 1997). The minimum area reserve selection method aims to represent all natural features (e.g. species or habitats) a given number of times in the smallest possible area, fewest numbers of sites, or with the lowest overall cost. Typically, analyses of this type have concentrated on the identification of the minimum set of sites required to represent all species at least once (see for example Rebelo and Siegfried 1992, Sætersdal et al. 1993, Kershaw et al. 1994, Lombard et al.

1995, Murkiuki et al. 1997, Pressey et al. 1997, Rodrigues et al. 2000b). By contrast, maximal coverage approaches aim to maximise the representation of natural features (e.g. species or habitats) given a limit to the number of sites, overall cost or area (Arthur et al. 1997, Pressey et al. 1997, Rodrigues et al. 2000a, Cabeza and Moilanen 2001).

It has been argued that a major constraint on the practical application of these exact algorithms is computational intensity (e.g. Pressey et al. 1996) as this increases substantially with the number of potential sites for inclusion. Indeed, this is often cited as reason enough to use sub-optimal heuristics. Nonetheless, following considerable advances in software and computational capabilities, only the largest problems, in terms of the size of the species/sites matrix, may prove to be intractable (Rodrigues and Gaston 2002).

Representing all species at least once in the minimum area (minimum set) is the most commonly applied site-selection objective (see for example Sætersdal et al. 1993, Kershaw et al. 1994, Parga et al. 1996, Williams et al. 1996, Csuti et al. 1997, Pressey et al. 1997, Rodrigues et al. 2000b). This objective can be written as:

$$\text{Minimise: } \sum_{j=1}^{n} x_j \qquad \text{(I)}$$

$$\text{Subject to: } \sum_{j=1}^{n} \alpha_{ij} x_j \geq 1 \qquad i = 1,2,\ldots\ldots\ldots, m \qquad \text{(II)}$$

$$x_j \in \{0,1\} \qquad j = 1,2,\ldots\ldots\ldots, n \qquad \text{(III)}$$

Where: m = the total number of species and n, the total number of sites; α_{ij} is 1 if species i is present in site j and 0 otherwise; and x_j is 1 if site j is selected and 0 otherwise (Csuti et al. 1997, Ando et al. 1998, Rodrigues et al. 1999, Rodrigues et al. 2000a, Cabeza and Moilanen 2001). The objective function (I) states that the minimum number of sites should be selected in order to satisfy the constraints (II) set out by the problem. In this instance, the constraint (II) is that all species must be represented in the resultant network of sites at least once. Finally, (III) states that each variable x_j is either 0 or 1, forcing each selection area to be treated as an indivisible unit (i.e. only whole sites can be selected). This basic formulation can then be modified to incorporate various design factors such as cost, area, representation targets and network connectivity depending on the particular objectives of individual conservation planning project (for details see Rodrigues et al. 2000a).

4.3 Methods

Data

Analyses were conducted for 17 species of waterbird using data from the Wetland Birds Survey (WeBS). Data for the years 1976/77 (earliest point where counts for each of these species was routinely collected) to 1998/99 (latest available data as of September 2001) inclusive were inlcuded for whooper swan, Bewick's swan, European white-fronted goose, dark-bellied brent goose, shelduck, wigeon, gadwall, teal, mallard, pintail, shoveler, pochard, tufted duck, goldeneye, smew, red-breasted merganser and goosander. These species are those considered well represented by WeBS counts and for which these are the primary source of data used for the selection of SPAs and Ramsar Sites in the UK.

The PS network is not static and new sites are classified in each year (see Fig. 2.1). Therefore, the numbers of SPAs and Ramsar Sites increased in each year of the analysis. To maintain a consistent number across all years the PS network for each year was taken as the 138 sites listed as SPAs and/or Ramsar Sites in the 1998/99 count year.

Networks

The performance of the existing (1998/99) PS network was compared with hypothetical networks selected by the following five alternative methods:

i) Generating one hundred random sets of sites, each comprising the number of sites equal to that in the actual PS network (138) selected from all WeBS sites;
ii) Maximising the total numbers of birds across all 17 species in the network in 1998/99, ensuring that for each year between 1994/95 and 1998/99 inclusive, population totals for each species were at least equal to that of the existing PS network;
iii) As in (ii) but using 1980/81-1984/85 data;
iv) Maximising the total numbers of birds across all species in the network for the 1998/99 count year, while ensuring that all species attain greater than a given representation target within the network as a whole for each year between 1994/95 and 1998/99 inclusive;
v) As (iv) but using 1980/81-1984/85 data.

For networks (ii) to (v) a five-year period of data was used to directly compare with the current PS selection process.

Representation targets for strategies (iv) and (v) were derived following the procedure outlined by Bezzel (1980), and modified by Stroud et al. (1990) and Jackson et al. (in press-b). Originally devised for use with breeding populations and advocated as a suitable means for identifying potential SPAs (Stroud et al. 1990), this index scores each species on four axes: a) species range within the western Palearctic (derived from Cramp and Simmons 1994); b) the proportion of the biogeographic population occurring in Great Britain (Jackson et al. in press-a; Chapter 2); c) wintering biogeographic population in pairs (Rose and Scott 1997); and d) long-term (1974-1996) population trends (Wetlands International 1996). Details of this method can be obtained from Stroud et al. (1990). Briefly, each species was awarded a score for each of the four variables, on a scale of 0 to 9:

a) 0 = a population range of >75.0% of the western Palearctic, 1 = >50.0%, 2 = >40.0%, 3 = >30.0%, 4 = >20.0%, 5 = >10.0%, 6 = >5.0%, 7 = >1.0%, 8 = >0.1%, 9 = <0.1%;

b) 0 = <9.9% of the biogeographic population in Great Britain, 1 = 10.0-19.9%, 2 = 20.0-29.9%, 3 = 30.0-39.9%, 4 = 40.0-49.9%, 5 = 50.0-59.9%, 6 = 60.0-69.9%, 7 = 70.0-79.9%, 8 = 80.0-89.9%, 9 = >90.0%;

c) 0 = >1,000,000 wintering pairs in the biogeographic population, 1 = >100,000, 2 = >50,000, 3 = >10,000, 4 = > 5,000, 5 = >1,000, 6 = >500, 7 = >100, 8 = >50, 9 = <50;

d) 3 = species has colonised since 1950 and is spreading, 4 = clear long-term increase, 5 = long-term feeble or patchy increase, 6 = long-term trend stationary, or no information available, 7 = long-term feeble or patchy decrease, 8 = long-term clear decrease, 9 = died out in the 20th century.

Table 4.1: The representation targets (%) calculated for each of the 17 species analysed for selection methods (iv) and (v) using the modified Bezel index. Given that networks derived using methods (iv) and (v) incorporate different windows of annual counts data, where appropriate (i.e. to determine the proportion of the biogeographic population occurring in Great Britain and the wintering biogeographic population) the corresponding years of data (1994/95-1998/99 for method (iv) and 1980/81-1984/85 for method (v) were used to determine the index values.

Species	**Method (iv)**	**Method (v)**
Whooper swan	>=80	>=80
Bewick's swan	>=80	>=80
European white fronted goose	>=20	>=20
Dark-bellied brent goose	>=40	>=40
Shelduck	>=20	>=40
Wigeon	>=20	>=20
Gadwall	>=20	>=40
Teal	>=20	>=20
Mallard	>=20	>=20
Pintail	>=40	>=40
Shoveler	>=20	>=40
Pochard	>=20	>=20
Tufted duck	>=20	>=20
Goldeneye	>=20	>=20
Smew	>=20	>=20
Red-breasted merganser	>=20	>=20
Goosander	>=20	>=20

Given that networks derived using methods (iv) and (v) incorporate different windows of annual counts data, as appropriate (i.e. to determine the proportion of the biogeographic population occurring in Great Britain (b) and the wintering biogeographic population (c)) the corresponding years of data were used to determine the index values.

The individual scores for a species from the four variables were summed to produce an overall Bezzel score. To convert these index values to representation targets, this value was divided by 36 (the maximum possible), converted to a percentage and allocated to one of four classes: >= 20%, >= 40%, >= 60% or >= 80% (Table 4.1). Given that Annex 1 species are priorities because of their vulnerability, these were automatically given a representation target of >= 80%.

Network assessment

To assess the performance of the existing (1998/99) PS network, in accordance with the methodology adopted by WeBS, peak annual counts for the 17 species, both individually and in aggregate, were calculated (see Pollitt et al. 2000). In each year (1976/77-1998/99) these were taken as the maximum count from the individual monthly counts across all selected WeBS sites in Great Britain. These national totals were then compared with the numbers of birds occurring in networks obtained using the five alternative selection methods in each year. In addition, for each of the species individually, network comparisons were made using the ratio between the numbers of birds held by methods (ii) to (v), and the numbers of each species on the actual PS network (termed the 'Gap') at four equally spaced (six year) intervals: 1980/81, 1986/87, 1992/93 and 1998/99.

Throughout, optimal solutions to network design problems were determined by means of a linear programming optimisation algorithm (see Rodrigues et al. 2000c for details of this procedure) using CPLEX™ (ILOG 1999). When solving the problems, all possible optimal solutions were found by sequentially excluding preceding optimal solutions by adding them as additional constraints to the algorithm. This ensured that the same combination of sites could not be re-selected. For the random selection method, the average numbers of birds across all 100 separate networks selected is presented as the solution for this method.

4.4 Results

All species

In conjunction with an overall increase in the total numbers of the 17 waterbird species recorded nationally between 1976/77 and 1998/99, the existing PS network (138 sites) similarly included an increasing total number of birds. Specifically, the PS network included from 360,656 birds in 1976/77 to 611,575 in 1998/99, an overall increase of 250,919 birds (Fig. 4.1). In addition, the total numbers of birds included in the PS tracked closely those of the national total over time, with coincident peaks in 1978/79, 1984/85, 1991/92 and 1996/97; the percentage of the national total within the PS network remained practically constant, with on average 62.9% ±2.02% (mean ±(SD)) of the total birds included annually.

The randomly generated networks consistently held the lowest total number of birds when compared with the PS network (Fig. 4.1). In all years the totals produced by this method comprised a very low proportion of the aggregate national total for the 17 species; for example, 6.2% of the national total for the 17 species in 1980/81 (*N* = 720,496) and 5.9% in 1998/99 (*N* = 976,322).

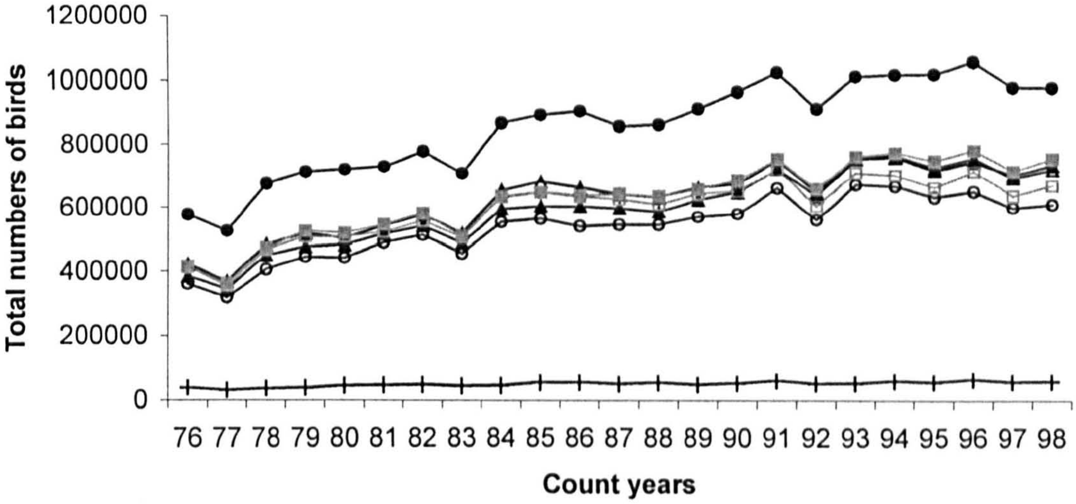

Figure 4.1: The total number of birds recorded for each year between 1976/77-1998/99 on all WeBS sites (•) compared with the numbers of birds in the existing PS network (○) and in networks of the same size selected using each of the five selection methods: i) +, ii) ▲, iii) Δ, iv) ■ and v) □.

Although the PS network contained a high proportion of the national total for these 17 species, the other alternative selection methods performed consistently better (Fig. 4.1). For example, compared to 60.3% of the national total included by the PS network in 1990/91, 67.3% was included using method (ii), 70.1% for method (iii), 71.6% for method (iv), and 67.8% for method (v). Similarly, in 1998/99 the PS network contained 62.6% of the national total, compared to 75.4% using method (ii), 74.0% for method (iii), 77.4% for method (iv) and 68.6% for method (v). Correspondingly, on average over the 23-year period, a greater percentage of the national total was included for each of these area selection methods ((mean ± (SD)) ii = 69.2% (±2.56), iii = 72.9% (±1.85), iv = 73.2% (±1.96), v = 69.8% (±2.28)). The small standard deviations mean that, like the PS network, these hypothetical networks were also robust over time, maintaining a relatively constant percentage of the national total relative to annual fluctuations.

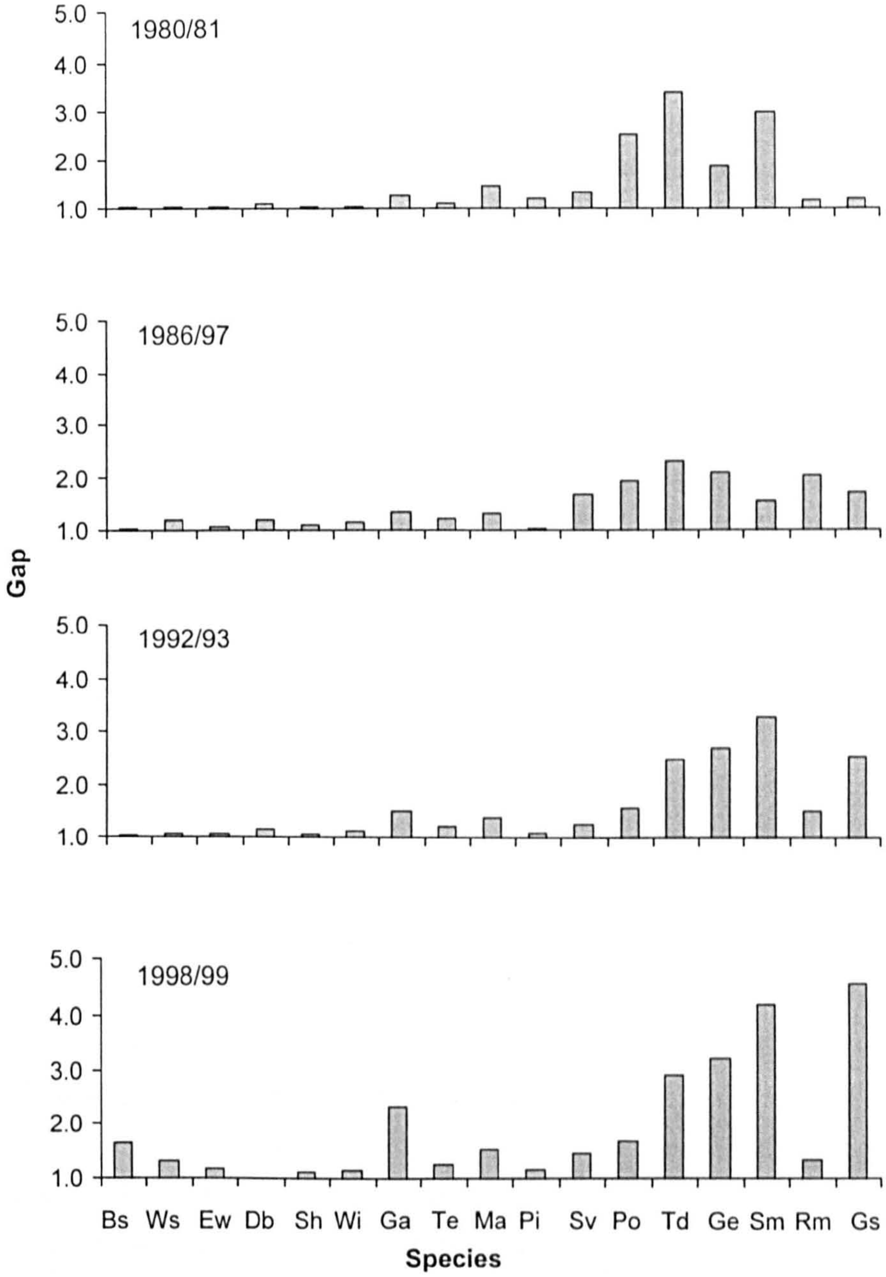

Figure 4.1: The gap (ratio between the numbers of birds included in networks using strategy (iv) compared with the numbers of birds on the existing PS network for each of the 17 species) taken at six year intervals: 1980/81; 1986/87; 1992/93; 1998/99. For gaps greater than one the existing PS network included fewer birds, and less than one more birds. The x-axis letters refer to each of the species: Bs) Bewick's swan, Ws) whooper swan, Ew) European white fronted goose, Db) dark-bellied brent goose, Sh) shelduck, Wi) wigeon, Ga) gadwall, Te) teal, Ma) mallard, Pi) pintail, Sv) shoveler, Po) pochard, Td) tufted duck, Ge) goldeneye, Sm) smew, Rm) red-breasted merganser, and Gs) goosander.

There were no significant differences between the numbers of birds included by each of the alternative strategies (ii to v). There were, however, significant differences between the numbers of birds in the existing PS network and each of strategies (ii) to (v) (comparisons with (ii) $t = -1.99$, *d.f.* = 22, $P < 0.05$; (iii) $t = -2.76$, *d.f.* = 22, $P < 0.01$; (iv) $t = -2.79$, *d.f.* = 22, $P < 0.01$; and (v) $t = -2.15$, *d.f.* = 22, $P < 0.01$).

Individual species

The PS network of 138 sites consistently included significantly greater numbers of birds for each individual species when compared with the random networks. In comparison with the hypothetical networks produced using strategies (ii) to (v), the individual species representation was almost always lower or at least equal for the PS. Indeed, this was always true for strategy (iv) (Fig. 4.2). In addition, for many of the 17 species the gap (ratio between the numbers of birds incorporated into networks selected using the alternative methods and those included in the actual PS network) typically increased over time.

The largest gaps were observed using 1998/99 data (e.g. Fig. 4.2). For example, the maximum number of goosander included by method (iv) was 4.6 times that of the PS network (Fig. 4.2). More specifically, method (iv) included 2,286 goosander in 1998/99 (64.6% of the national population, $N = 3{,}536$) compared with 502 birds (14.2% of the national total) recorded on the PS network. Similarly method (ii) incorporated 4.5 times more smew, approximately 78% of the national total ($N = 350$) in 1998/99, compared with 61 (17.4% of the national total) in the actual PS network (Fig. 4.2).

The gap between the numbers of birds on sites selected using the alternative selection methods and those on the PS network was greater for those species with a more dispersed distribution pattern compared with aggregating species (Fig. 4.3a-f and Fig. 4.4). For example, in all years substantially more birds were included on networks selected using the four alternative methods for goldeneye (Fig. 4.3a), smew (Fig. 4.3b) and goosander (Fig. 4.3c), three dispersed distribution species, as demonstrated by the larger gaps.

In contrast, for European white-fronted goose (Fig. 4.3d), dark-bellied brent goose (Fig. 4.3e) and shelduck (Fig. 4.3f), three species with an aggregated distribution across suitable wintering sites, the differences between the actual PS network and the alternative methods in relation to the numbers of birds included was minimal for all years, with approximately the same numbers of birds included in all networks. Indeed, in comparison with method (ii), the PS network on occasion contained slightly more dark-bellied brent geese (Fig. 4.3e and Fig. 4.4). However, in all years methods (iii), (iv) and (v) included greater, or at least equal, numbers of dark-bellied bent goose compared with the PS network.

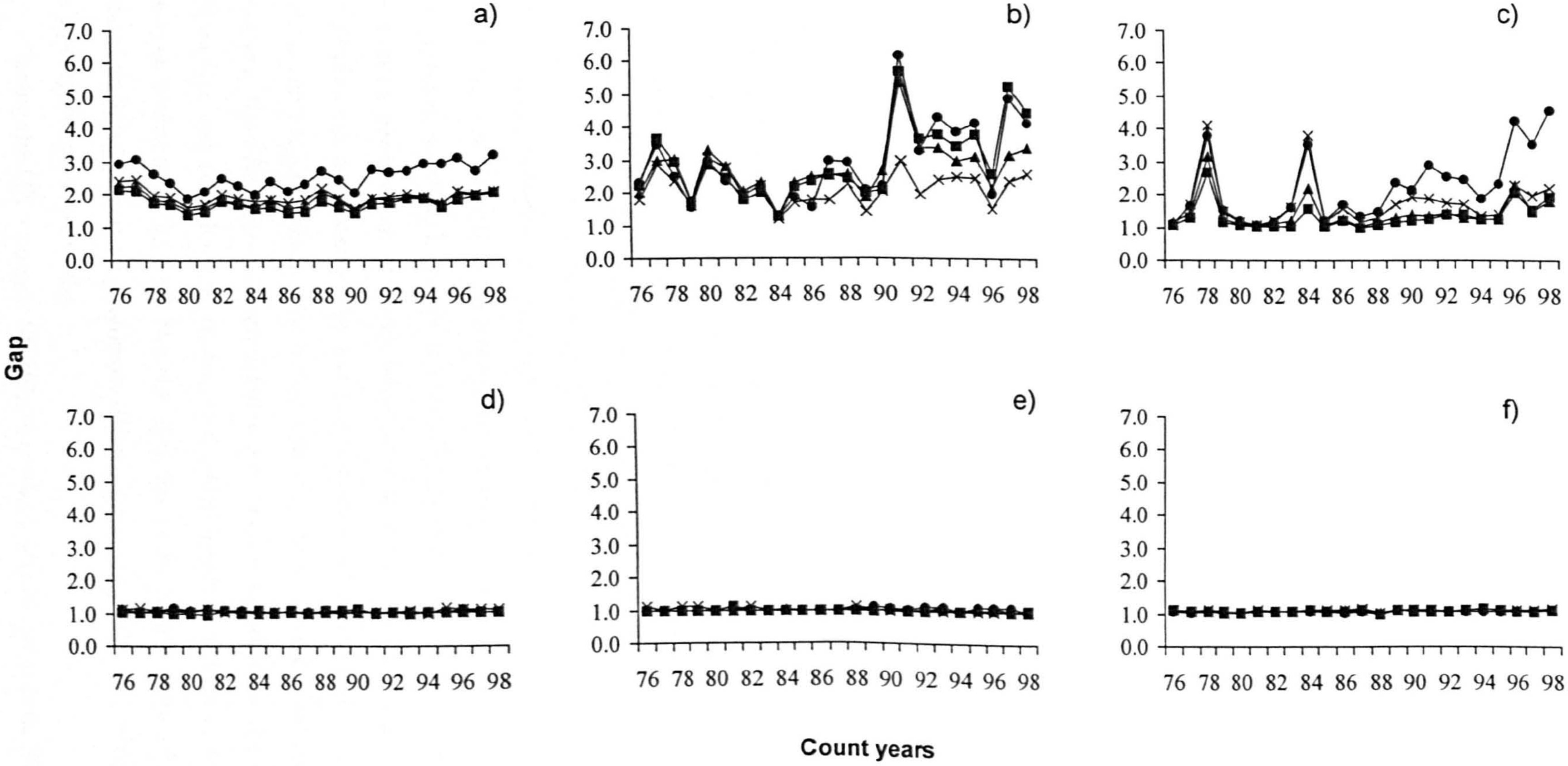

Figure 4.3: The gap (ratio between the numbers of birds included on networks selected using strategies (ii) (■), (iii) (▲), (iv) (●) and (v) (X) compared with the numbers on the existing PS network) for a) goldeneye, b) smew, c) goosander, d) European white-fronted goose, e) dark-bellied brent goose and f) shelduck between 1976/77 and 1998/99.

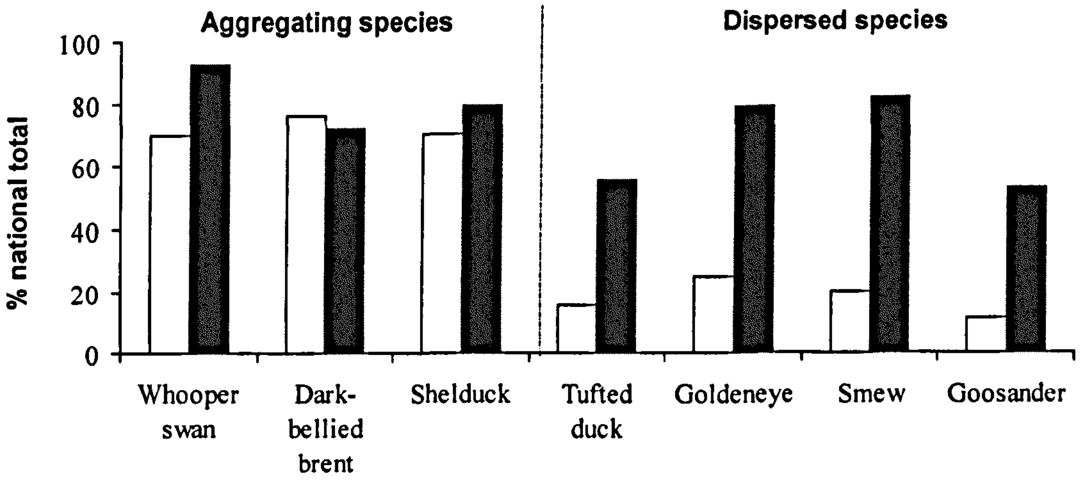

Figure 4.4: The percentage of the national population occurring within the existing PS network (open bars) compared with population totals for sites selected using method (iv) (filled bars) for a selection of the 17 species analysed.

4.5 Discussion

Given the inherent complexities of site-selection, prospective sites for inclusion in national-scale conservation networks are generally evaluated in isolation. Such methods have undoubtedly provided many suitable sites for the protection of important waterbird populations in the region. Encouragingly, the existing PS network was shown to be especially robust over time, successfully supporting approximately 60% of the total numbers of wintering waterbirds in all 17 years. However, despite this excellent performance in total population conservation, such an approach may not be the most effective means of selecting potential sites with respect to the representation of each species individually (as shown in Chapters 2 and 3). Indeed, from a conservation perspective, although total numbers are necessarily important in general terms, their attainment is secondary to the representation of individual target species within the network (particularly since this is the specific legal objective of Article 4 of the Birds Directive). Specifically, when compared to the network-based methods (ii to v), the existing PS network was less effective with respect to the total numbers of each of the 17 species analysed here in each year (see Fig. 4.1). Not only this, but, in the same number of sites, these alternative methods included greater total numbers of birds (approximately 70% of the national total) in the resultant hypothetical networks.

Using the UK's current SPA/Ramsar Site selection procedure, the degree to which the 17 waterbird species are represented in the PS network is largely a function of species dispersal and distribution in Great Britain (discussed in Chapters 2 and 3). In particular, those species that tend to have large aggregated populations will be present as sizeable proportions of their national totals within the existing PS network. For example, for European white-fronted goose and dark-bellied brent goose, which typically exhibit a clumped population distribution, the gap between numbers included by the alternative methods and on the actual network was far smaller than for dispersed distribution species such as goldeneye and goosander. This was not unexpected given that originally, the threshold of 1% of either the national total (Annex I) or biogeographic population (regularly occurring migratory species) for distinguishing important sites was developed as a means of targeting those species that tend to concentrate on a few key sites (Atkinson-Willes et al. 1982). Thus, the use of this criterion will result in high proportions of such species being included in the PS network, and it is assumed that dispersed species will benefit indirectly from protected areas designated or classified for other species (Atkinson-Willes et al. 1982). Although this may seem logical, for several dispersed distribution species the proportions of the national and in particular the international populations afforded protection in Great Britain by the PS are considerably lower than those of the aggregating species (discussed in Chapters 2 and 3). Given their wide-ranging distribution, many such species are not of immediate concern to conservationists, some are. Several dispersed distribution species are included in the JNCC 'Birds of Conservation Concern 2002-2007' as amber list species (those of medium conservation concern), including shoveler, pochard and goldeneye (JNCC 2002). Additionally, as discussed in Chapter 2, those species that are widespread and abundant today may be species in great need of conservation in the future. Moreover, systems such as wetlands alter over time both with natural succession and due to anthropogenically induced interference. Consequently, if all such species are well represented within the PS network any unexpected alterations in the degree of threat and/or conservation status may be more easily countered (Hopkinson et al. 2000b).

In contrast to the existing PS network, the incorporation of species-specific representation targets (as either percentages or baseline numerical population targets) into the site-selection procedure, as suggested by Stroud et al. (1990), ensured that all species at least attained some minimum level of representation. In particular, the percentage targets calculated using the modified Bezzel index explicitly incorporate measures of a species' national and international importance, based on the distribution patterns, population size and long-term population trends. Consequently, the level of representation is related to the individual requirements for each species, and is not simply a function of total numbers at a given site. In the same number of sites as the existing PS network, greater numbers of dispersed species such as goldeneye, goosander and smew were represented by networks (iii), (iv) and (v), with no associated compromise for numbers of aggregating species (shown for method (iv) in Fig. 4). For example, the numbers of smew were increased from less than 20% of the national total in the existing SPA network to approximately 85% using sites selected using method (iv) in 1998/99. Similarly, numbers of Bewick's swan were increased by approximately 45% using this same method (Fig. 4.4). This lack of compromise regarding the numbers of aggregating species is not, however, observed for all methods. For European white-fronted goose on occasion there was a decrease in the numbers included on networks selected using method (ii) compared with the PS network. This demonstrates firstly the success of the current PS network in providing sites for this species, and secondly, that in order to provide for all species adequately, methods based on representation targets ((iv) and (v)) ultimately perform better over time. Of the methods used here, (iv) was shown to identify those sites where the populations of all 17 species, both individually and in aggregate, were maintained over time in the largest numbers (on average approximately 73% of the national total across the 17 species).

4.6 Network improvement

The performance of the existing SPA network can be improved, in terms of the numbers of wintering waterbirds included, if the future SPA capability (82 wetland sites proposed and/or classified as SPAs after 1998/99) is taken into account (Table 4.2 and Fig. 4.5). Indeed, the addition of these 82 sites to the 138 listed in the 1998/99 SPA network increases the total numbers of birds protected from 611,753 to 663,476 birds in 1998/99 (i.e. by 8.4%). However, a still greater improvement could have been achieved by adding just ten sites (four of which are included in the future SPA network), identified using the four alternative selection methodologies detailed in this analysis (Table 4.2 and Fig. 4.6). In 1998/99, these ten sites would increase the total numbers of the 17 species within the SPA network by 16.5%, from 57.6% to 72.8% of the national total. In addition, for each of the 17 species the individual species representation was greater for the 148 sites network (138 sites plus the top ten non-SPA sites) compared with the 220 sites future network (Table 4.2).

4.7 Implications and conclusions

Although the existing PS network performed extremely well in terms of the numbers of birds included compared with hypothetical networks selected using a random approach, these analyses suggest the current approach to site-selection is not the most effective in terms of individual species representation. Indeed, in the same number of sites as for the existing PS network, focussing on complementarity between component sites rather than applying criteria to each site individually, not only improved the level of representation for each species, but also included a greater proportion (approximately 70%) of the national total across all 17 species and was robust over time.

In light of these findings, explicit changes regarding the aims and objectives of PS for waterbirds and also the means by which sites are considered during the selection process might usefully be recommended, both at national and international level. I suggest a modified approach for the selection of future PS, where sites are selected relative to their contribution to the network as a whole and where all species for which protected areas are required under the

Ramsar Convention and Article 4 of the Birds Directive are given ecologically sound representation targets further to ensure their persistence in the long-term. Following such a procedure may assist in the realisation of the objectives of the Birds Directive and the Ramsar Convention regarding the protection of waterbird populations in the UK; particularly the achievement of the aims of Article 4, which states that the selected sites should 'form a coherent whole which meets the protection requirements of these (target) species'.

Table 4.2: The 1998/99 population totals for: a) the current PS network (138 sites), b) the future network (138 sites plus 82 proposed or recently classified sites), c) the 148 sites network (138 sites of the existing network, plus the top ten non-PS), and the % difference between the future and 148 sites networks for each of the 17 species analysed.

	Numbers of birds			
Species	a	b	c	Difference (%)
Whooper swan	2693	2824	2957	4.71
Bewick's swan	3880	3886	4159	7.03
European white-fronted goose	3872	3878	4133	6.58
Dark-bellied brent goose	73354	73381	73996	0.84
Shelduck	51186	58247	64822	11.29
Wigeon	280916	293944	321191	9.27
Gadwall	3644	5652	6642	17.52
Teal	87462	95985	98986	3.13
Mallard	48589	54946	58958	7.30
Pintail	21414	25590	26946	5.30
Shoveler	3768	4590	4929	7.39
Pochard	13804	15260	17072	11.87
Tufted duck	10114	14211	15682	10.35
Goldeneye	4106	7110	7479	5.19
Smew	61	133	145	9.02
Red-breasted merganser	2388	3259	3663	12.40
Goosander	502	580	764	31.72
Totals	611753	663476	710950	7.16

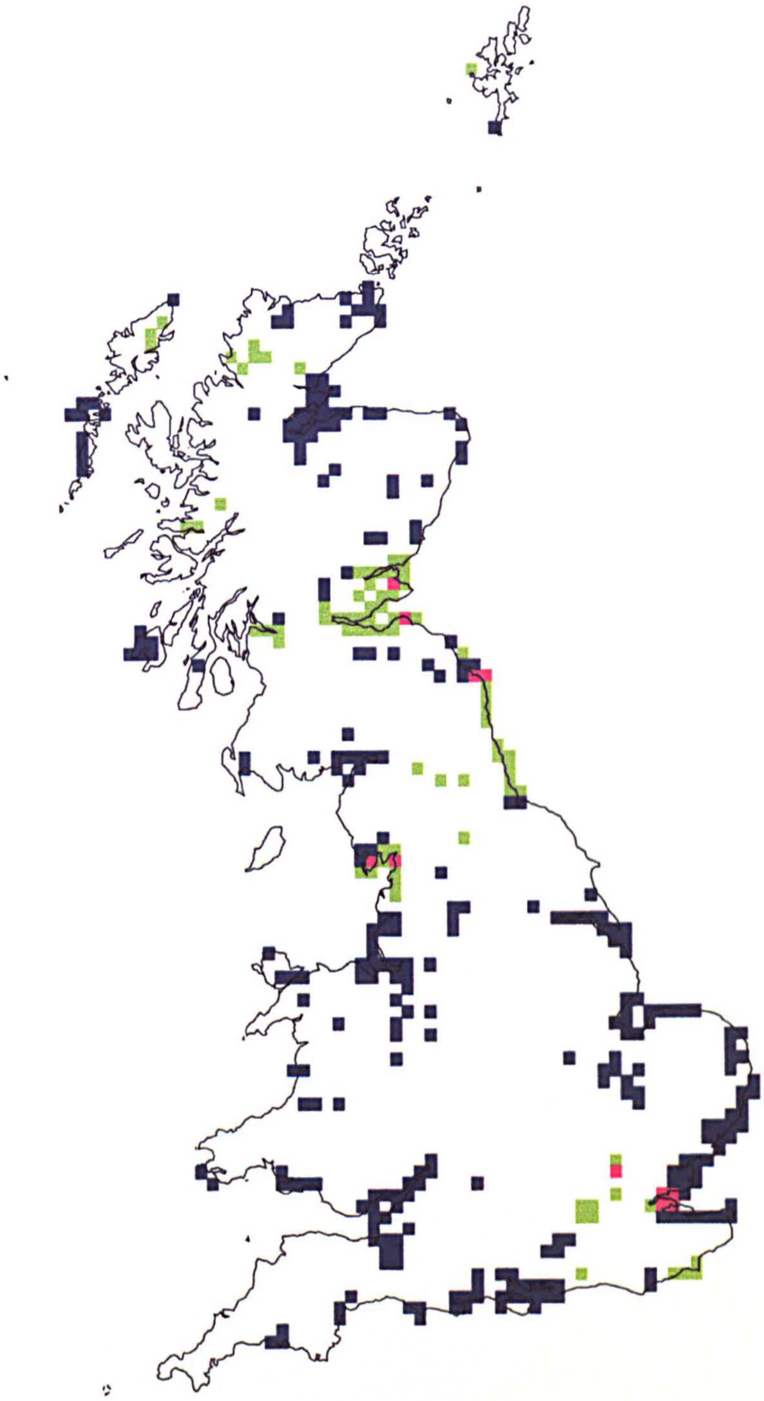

Figure 4.5: Location of all existing (navy squares), future protected areas (red squares) and extended protected areas (green squares).

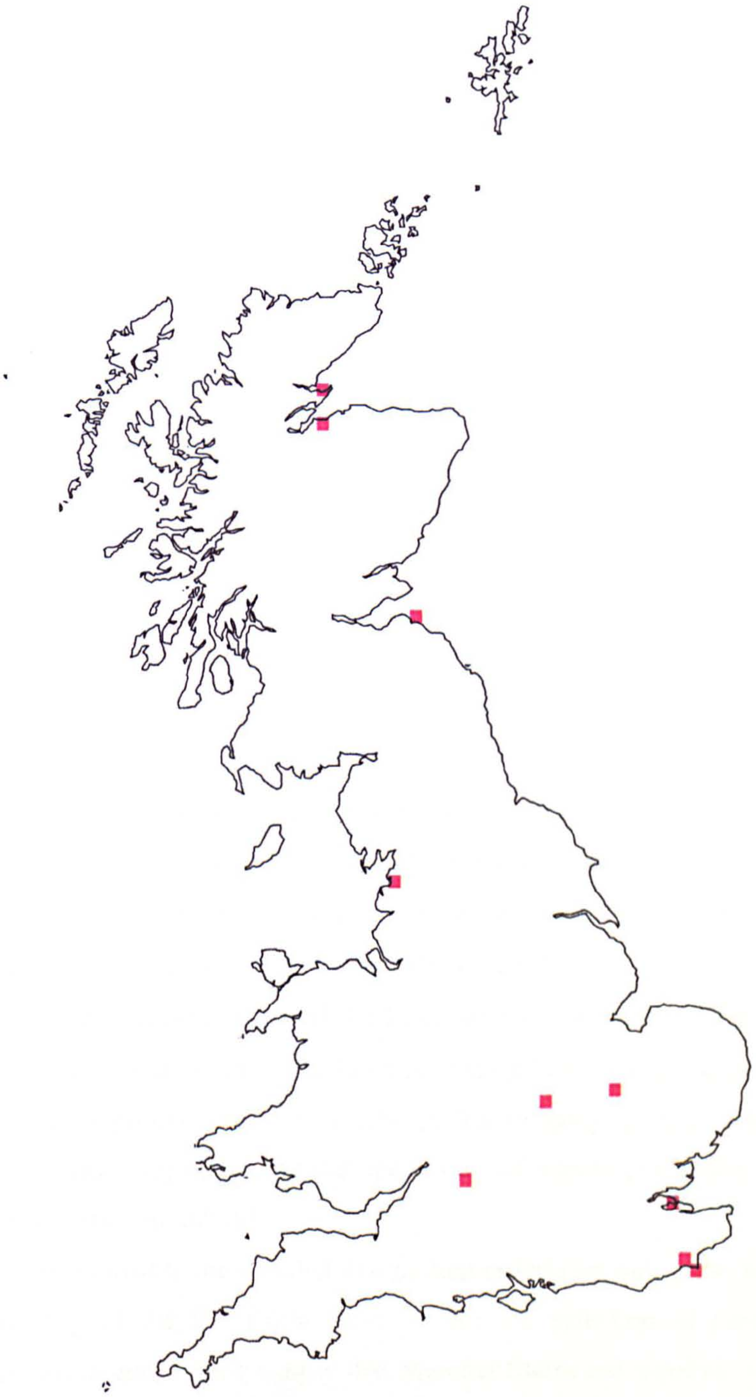

Figure 4.6: Location of the ten additional wetland sites required to increase the representation of each of the 17 species individually and in total.

CHAPTER 5: NETWORK IMPROVEMENTS

5.1 Identifying priority areas for waterbird conservation: Data types and windows

5.1 Introduction

Growing concerns as to the status of many existing protected areas and increasing pressures on land for anthropogenic development have led to a drive towards more rigorous priority area selection methods. Despite their obvious advantages in terms of safeguarding biodiversity, such as increased efficiency and effectiveness of reserve networks, many such methods focus on the attainment of numerical threshold criteria and therefore rely to a large extent on the characteristics of the biological information available to planners (Nicholls and Margules 1993, Pressey et al. 1993, Freitag et al. 1996, Haila and Margules 1996, Willis et al. 1996, Freitag et al. 1998, Freitag and van Jaarsveld 1998, Cabeza and Moilanen 2001, Williams et al. 2002). Indeed, the quality, type and quantity of input data ultimately will strongly influence how well chosen sites represent target populations. In this respect, signatories to the Ramsar Convention on Wetlands of International Importance are expected to select internationally important wetlands for waterbirds using numerical thresholds for population size (for details see Ramsar Convention Bureau 2002b). Specifically, of the eight designation criteria detailed in Article 2.2 of the Ramsar Convention, as of Feb. 2002, 23% of all Ramsar sites have been designated on the basis of criteria 5 and 6 where a site is considered important if it regularly supports greater than 1% of the biogeographic population (usually that of the western Palaearctic) of a particular species or sub-species, or 20,000 birds (Article 2.2; Ramsar Convention Bureau 2002b).

In contrast to the detailed designation criteria set out by the Ramsar Convention Bureau, the wording of the EU Birds Directive for the selection of potential SPAs has been left deliberately vague, stating simply that Member States are required 'to identify and designate as Special Protection Areas the most suitable territories in number and size' (see Chapter 2). Although it has been argued that a state reluctant to classify SPAs could use this generality to their advantage (Owen 2001), this means that individual Member States are free to administer selection methodology at their own discretion, provided the overall objectives of the Directive are met. In Great Britain, there is a great deal of overlap between the selection criteria specified by the Ramsar Convention and those adopted by the JNCC for the selection of Special

Protection Areas (SPAs) (see Chapter 2). Briefly, a site is considered for inclusion in the SPA network if it supports 1% of the biogeographic population of all species listed in Annex I of the Birds Directive, or 1% of the national (GB or all-Ireland) population of all species listed as Annex II (Stroud et al. 2001).

In light of the lack of explicit criteria for the selection of SPAs within the Birds Directive, the adoption of disparate selection methods for SPAs amongst EU countries will likely influence the effectiveness of resultant network performance. Indeed, more generally the choice of site selection method has previously been shown significantly to affect the outcome of reserve selection and how well chosen sites represent target populations (see for example, Kershaw et al. 1994, Csuti et al. 1997, Virolainen et al. 1999). However, although each of these studies looked at the implications of alterations made to the overall site selection method on the effectiveness of protected area networks (for example, different types of heuristic site selection algorithms; Csuti et al. 1997), none addressed the implications of changing the characteristics of the data themselves. Using waterbirds in the UK as a case study, in this Chapter I assess the effect of alterations made to the amount (the number of years of data) and format (mean counts or peak counts) of abundance data incorporated into a selection process, and the numbers of sites to be selected, on the performance of hypothetical reserve networks selected using a linear programming selection algorithm. The findings are of interest more widely than the bounds of the EU, as they bear on the criteria for the selection of priority areas for waterbirds elsewhere.

5.2 Methods

Data

Data for the years 1981/82 to 1990/91 inclusive were included, for Bewick's swan, whooper swan, European white-fronted goose, dark-bellied brent goose, shelduck, wigeon, gadwall, teal, mallard, pintail, shoveler, pochard, tufted duck, goldeneye, smew, red-breasted merganser and goosander. These species are those considered well represented by WeBS counts and for which these are the primary source of data used for the selection of SPAs in the UK. For these analyses 1962 sites were included, the same set for each of the 17 species and in all years.

Individual site counts

The total bird count at each individual wetland site was calculated for each of the 17 species individually and for the 17 species summed for each year (1980/81-1989/90). In line with the methodology adopted by WeBS (see Pollitt et al., 2000), individual site counts for each year were taken as the maximum of the individual monthly counts from September to March. These counts relate to the total numbers of birds recorded on each distinct wetland area and will, therefore, represent aggregate totals for one or more of the individual WeBS counting units (see Chapter 1).

Network selection algorithms

The site-selection approach used in this paper is a modified maximal coverage problem (Arthur et al. 1997; Chapter 4). Originally developed for operations research, these mathematical reserve selection methods aim to maximise the representation of natural features (e.g. species or habitats) given a limit to the number of sites, overall cost or area (Arthur et al. 1997, Pressey et al. 1997, Rodrigues et al. 2000a, Cabeza and Moilanen 2001). However, rather than simply attempting to represent all species a given number of times as has been the tradition for these types of analysis (e.g. McKenzie et al. 1989, Sætersdal et al. 1993, Pressey et al. 1994, Price et al. 1995, Williams and Revelle 1996, Csuti et al. 1997, Polasky et al. 2000, Rodrigues et al. 2000b), each algorithm used here selects from all possible sites (in this case $N = 1962$) the network that, in a given number of sites, maximises the total number of individual birds included and satisfies numerical representation targets calculated for each of the 17 waterbird species individually.

Representation targets for each of the 17 species were derived following the procedure outlined in Chapter 4 (Bezzel 1980; Stroud et al. 1990; Jackson et al. in press).

To determine the effects of the choice of priority site selection method and the characteristics of the data themselves on resultant network performance, two types of abundance data were used within the site-selection algorithm framework: a) mean counts and b) peak counts. Thus, for each individual wetland site, the numbers of birds included in the linear programming algorithm related to either the mean or peak numbers calculated over a set number of years (two to ten).

To determine the implications of changing the actual amount of information employed, using each data format (mean or peak counts), nine separate algorithms were constructed. These incorporated either the mean or peak number of individuals calculated using from two to ten years of abundance data. Thus, for example, where two years of abundance data were included, each species-specific constraint was formulated using the mean or peak numbers of individuals recorded between 1989/90-1990/91. In addition, for each of the nine separate algorithms, all combinations of years over the period 1980/81-1989/90 (two to ten year sets of data) were included as separate runs of the algorithm. Specifically, using two years of data there were 45 combinations of the ten years of data, 120 using three, 210 using four, 252 using five, 210 using six, 120 using seven, 45 using eight, ten using nine and one combination using all ten years of abundance data.

Thus far, the algorithms described would simply select all sites for inclusion in a network as this will inevitably satisfy the selection criteria and maximise the total numbers of birds. To determine the effects of differing limitations on the numbers of sites selected, for each data format (mean or peak) and for each data period (two to ten years of abundance data), the numbers of sites to be selected was set from 50 to 1000 (maximum representation of all species achieved) at intervals of 25 sites.

Table 5.1: Representation targets (% of the national total) calculated using the modified Bezzel index for each of the 17 species of waterbird individually.

Species	% Target
Whooper swan	>=80
Bewick's swan	>=80
European white fronted goose	>=20
Dark-bellied brent goose	>=40
Shelduck	>=40
Mallard	>=20
Gadwall	>=40
Pintail	>=40
Shoveler	>=40
Wigeon	>=20
Teal	>=20
Pochard	>=20
Tufted duck	>=20
Goldeneye	>=20
Smew	>=20
Goosander	>=20
Red-breasted merganser	>=20

Overall, therefore, 39 algorithms were constructed for each data period, 351 algorithms for each data format (mean or peak counts), giving a total of 702 separate algorithms. Throughout, the optimal solution to all algorithms was determined using CPLEX™ linear programming software (ILOG 1999). Each solution was evaluated using WeBS data for the count year 1998/99. Thus, for each hypothetical reserve network, the total numbers of birds was calculated across all selected sites using the sum of the individual site counts for 1998/99.

Given the numbers of species (17) and sites (1962) included in the analysis, multiple optimal solutions to each algorithm are inevitable (see Chapter 6 for discussion). However, as all optimal solutions will have the same numbers of sites and the same solution value in terms of the total numbers of birds included, for the purposes of these analyses only one such solution was obtained in each case.

5.3 Results

Solutions were feasible (i.e. the species representation targets could be met) for all 351 algorithms using the peak abundance counts (39,507 networks in total), but for only 344 using the mean counts (38,584 networks in total). Using this latter method, solutions were not possible where the number of sites to be selected was 50 and greater than three years of abundance data were included in the algorithm.

Table 5.2: Results of a 3-way ANOVA without replication showing the effects of each main factor (data format, sites and years) and each of the 2-way interactions. Method refers to either mean or peak abundance data, sites is the number of sites selected (50 to 1000) and years is the number of years of abundance data included in the algorithms (two to ten).

Source of Variation	d.f.	F	P
Data format	1	136832.90	0.0001
Sites	38	611.96	0.0001
Years	8	0.68	> 0.05
Data format * Sites	38	57.13	0.0001
Data format * Years	8	2335.39	0.0001
Sites * Years	304	0.96	> 0.05
Error	297		
Total	695		
Corrected Total	694		

Overall, 3-factor Analysis of Variance showed that there was a significant effect of the data format ($F_{1, 78090}$ = 136832.90; $P < 0.0001$) and the number of sites to be selected ($F_{38, 78090}$ = 611.96; $P < 0.0001$) on the total numbers of birds included in the resultant networks (Table 5.2). More specifically, in all cases greater numbers of birds were included in networks selected using the peak counts compared with the mean counts ($Z = -22.02$; $N = 78{,}091$; $P < 0.0001$; Fig. 5.1). Overall across the 351 site-selection algorithms for each method, using mean abundance data included on average 547,329 fewer birds compared with using the peak counts (1,101,887 (±113,658) birds (mean (±SD)) compared with 1,649,227 (±271,510) birds using peak counts).

There was a significant positive correlation between the numbers of sites to be selected and the total numbers of birds included in the hypothetical reserve networks using both the mean ($r_s = 0.401$; $N = 39{,}507$; $P < 0.0001$) and the peak counts ($r_s = 0.512$; $N = 38{,}584$; $P < 0.0001$).

For both methods, the numbers of birds included increased as the numbers of sites to be selected was increased (Fig. 5.1). However, there were always greater numbers of birds included using the peak counts compared with networks constructed using the mean counts. For example, 1,186,447 (±39,686) birds were included in networks selected using mean counts compared with 1,839,065 (±290,892) birds using peak counts when the number of sites was set to 1000. Using mean counts, to include 25% of the maximum numbers of birds (1,839,065

(±290,892)) required only 26 sites. Similarly, approximately 50 sites would be required to support 50% and 114 sites to support 75% of this maximum number.

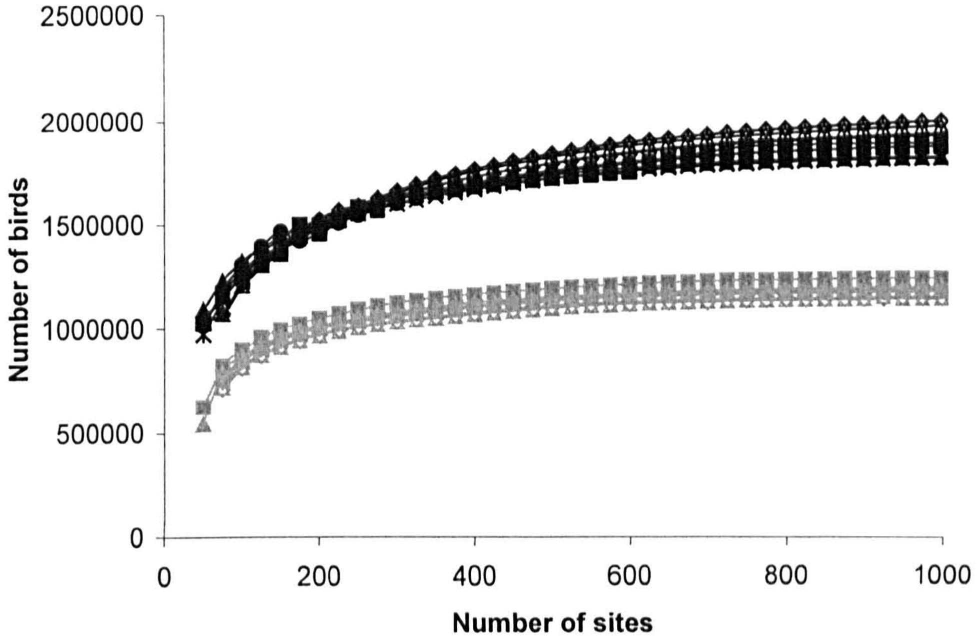

Figure 5.1: The numbers of birds included in hypothetical protected area networks selected using 702 site-selection algorithms where the numbers of sites to be selected increased from 50 to 1000. Black symbols relate to peak abundance counts and grey to means. Symbols relate to the numbers of years of abundance data incorporated into the algorithm: ■ = 2 years, ▲ = 3 years, x = 4 years, ⋆ = 5 years, ● = 6 years, + = 7 years, - = 8 years, Δ = 9 years, ◊ = 10 years.

In addition, considering hypothetical networks created using peak counts, approximately 11 sites were required to include 25% of the maximum numbers of individuals, approximately 49 sites to include 50% and approximately 117 sites were required to include 75% of individuals (calculated using S-curve estimation, SPSS version 11.0). Considering networks selected using both mean counts and peaks of abundance together, 50% of the total numbers of birds could be supported in less than 5% of the sites. Further, 75% could be supported in approximately 14% of the sites.

By contrast, there was no significant effect of the number of years of abundance data included ($F_{8,\ 78090} = 0.68$; $P > 0.05$) on the numbers of birds included in the hypothetical reserve networks (Table 5.2). Alterations to the numbers of years of data used in the basic maximal coverage algorithm (two to ten) made little difference to the effectiveness of the resultant hypothetical reserve networks using the peak abundance method (Fig. 5.3). Indeed, using peak counts, there was no significant correlation between the numbers of years of abundance data used in the algorithm and the numbers of birds occurring in the optimal networks for the count year 1998/99 ($r_s = 0.231$; $N = 39{,}507$; $P > 0.05$).

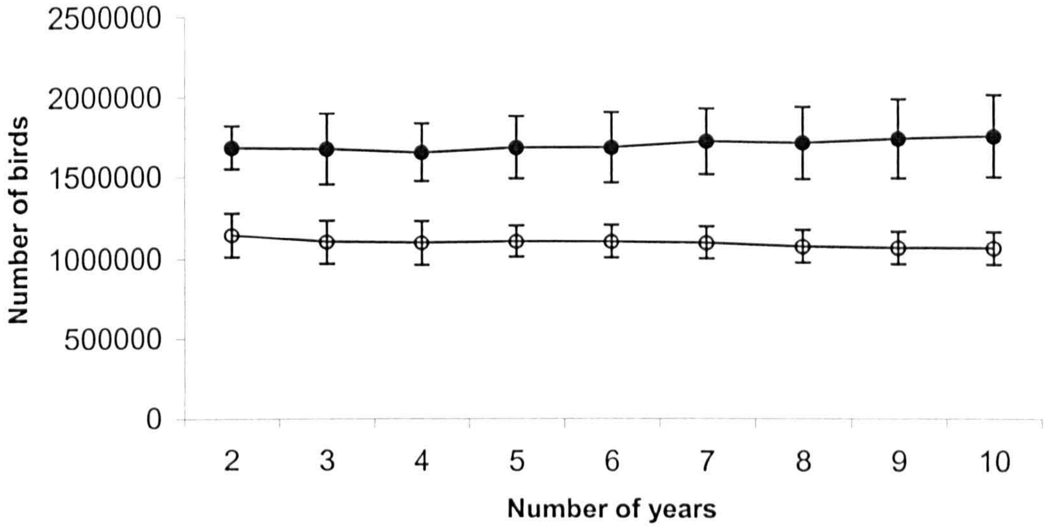

Figure 5.1: The mean numbers of birds included in hypothetical protected area networks selected using each site-selection algorithm, dependent on the size of the data window included (numbers of years of abundance data) for peak (●) and mean (○) abundance data. Mean values are calculated across all 39 algorithms for each year group (over 50 to 1000 sites). Error bars relate to the standard deviation of these means.

By contrast, using mean counts there was a significant negative correlation (r_s = -0.251; N = 38,584; $P < 0.0001$) between the size of the data window and the numbers of birds included in the hypothetical reserve networks. Here, the numbers of birds in the optimal solution decreased as the data window increased, from 1,164,873 (±133,481) individuals using two years to 1,062,424 (±99,465) using ten, although this equates to a decrease of only 8.8%.

There were also significant interactions between data format*sites ($F_{38,\ 78090}$ = 57.13; $P < 0.0001$) and data format*years ($F_{8,\ 78090}$ = 2335.39; $P < 0.0001$) (Table 5.2; Fig. 5.2). However, there was no significant interaction between sites*years ($F_{304,\ 78090}$ = 0.961; $P > 0.05$). Nonetheless, using peak abundance data only, there was a significant positive correlation between the standard deviation in numbers of birds calculated across the ten networks (two to ten years of data employed for each number of sites) and the numbers of sites selected (r_s = 0.999; N = 39; $P < 0.0001$). Thus, as the numbers of sites to be selected and the numbers of years of abundance data used increased, the numbers of birds included in the resultant networks also increased (Fig. 5.4). For example, using peak counts, the standard deviation in the numbers of birds increased from ±120,839 birds using 50 sites, to ±167,379 birds for 325 sites, to ±210,892 birds using 1000 sites. In addition, although there were always greater numbers of birds included in networks using ten years as opposed to two years of abundance data, there were significantly more birds included using ten years of data where the numbers of selected sites was greater compared with when fewer sites were selected.

By contrast, using mean counts, although there was a slight decrease in the standard deviation in the numbers of birds as the numbers of sites used increased, there was no significant correlation (r_s = -0.264; N = 39; P > 0.05). Thus, as the numbers of sites selected increased there was no significant effect of increasing the numbers of years of abundance data in the analysis.

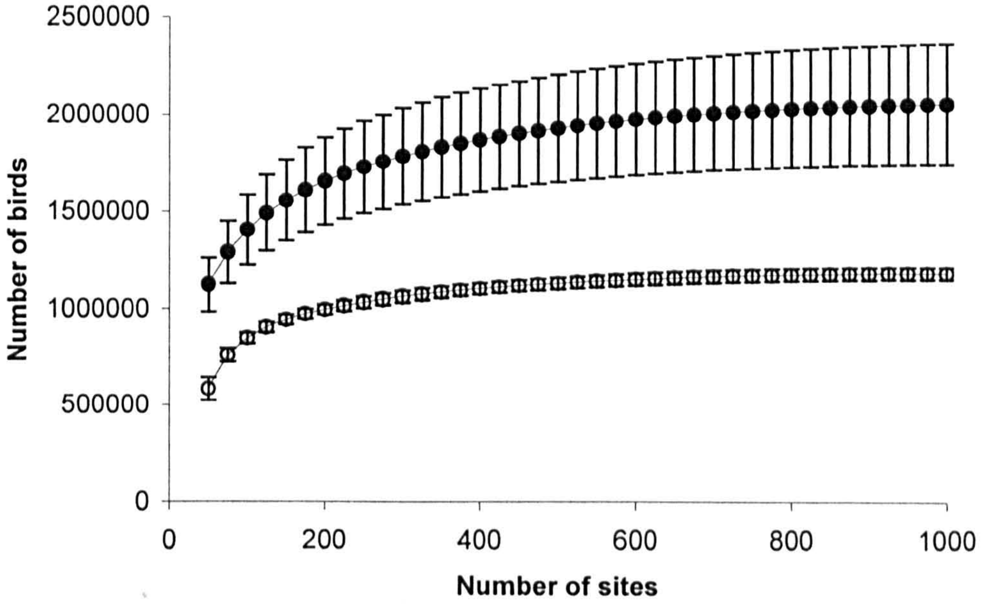

Figure 5.3: The mean numbers of birds included in hypothetical protected area networks selected using each site-selection algorithm, dependent on the numbers of sites selected (50 to 1000) for peak (●) and mean (○) abundance data. Mean values are calculated across all nine algorithms for each year group (two-ten years). Error bars relate to the standard deviation of these means.

Finally, as the number of years of abundance data included in the algorithm was increased from two to ten years the difference in optimal solutions, calculated for the 39 solutions (50 to 1000 sites) for each year of data, the difference in the numbers of individuals included between networks selected using mean and peak abundance data increased significantly. Indeed, the difference between the two methods in terms of the numbers of birds increased linearly with the number of years of data included (r_s = 0.867; N = 9; P < 0.0001).

5.4 Discussion

Selection methodology

Data inadequacies, coupled with pressure from competing land uses, are an inevitability for conservation planning. It is vital, therefore, that the most appropriate priority site-selection methods are chosen, not only to make the best use of the data available, but also so that the resultant priority-site networks are as representative as possible of a country's biodiversity.

Aside from issues of data quality/availability, it is clear from these analyses that the precise means by which priority sites are selected will undoubtedly impinge on resultant network effectiveness. Specifically, for these migratory waterbird species, using peak rather than mean counts across years always resulted in more effective networks. Indeed, the most effective approach for the selection of priority sites for the 17 species analysed incorporated peak rather than mean counts across years and the maximum possible number of WeBS sites.

Data considerations

The relative success or effectiveness of priority-site selection procedures for biodiversity conservation is fundamentally dependent on the quality, quantity and availability of biological survey data (Cabeza and Moilanen 2001). Ideally, all regions and species within a given dataset require equal sampling effort to enable the systematic comparison of potential areas for reservation (Margules et al. 2002). Unfortunately, in many cases, a lack of consistent, accurate and indeed comprehensive biogeographic data precludes accurate and informed decisions regarding conservation planning and reserve selection (Belbin 1993, Nicholls and Margules 1993). In reality, many data sets are far from ideal (for a review see Williams et al. 2002). For example, existing biological survey data are often compiled from a variety of sources, each with their own particular biases; for instance, field records are regularly taken opportunistically from areas where a species is known to exist; data are not always freely available for conservation planning; records are collected on an *ad hoc* basis by volunteers from places and times of their own choosing; data are often not collected for conservation purposes; data are generally presence only; and coverage is often scant and irregular (Hopkinson et al. 2000a, Hopkinson et al. 2000b, Polasky et al. 2000, Cabeza and Moilanen 2001, Williams et al. 2002). Indeed, the majority of published site-selection analyses highlight the confounding effects of data deficiencies (e.g. Khan et al. 1997, Araújo and Williams 2000). Further, although it is acknowledged that supplementary data collection would be undeniably beneficial for conservation planning exercises, collection is generally confounded by the lack of sufficient personnel, time and funds (Ehrlich 1992, Williams et al. 2002).

Accepting the global paucity of high quality biological datasets, the results reported in this analysis suggest that for the purposes of priority site selection there is limited benefit to carrying out systematic data collection as often as is the case for WeBS in the UK. Indeed, considering the results of networks using mean counts, greater numbers of years of abundance data incorporated into the site-selection framework significantly decreased the numbers of birds included in the resultant hypothetical networks. Thus, including ten years of abundance data was less effective in capturing numbers of the target species than a single year. In terms of the current approach to SPA selection adopted by the UK, the use of a single year of abundance records to represent the numbers of individuals at a site will likely be more effective in terms of network performance compared with the traditional five-year means approach. Additionally, considering the hypothetical reserve networks selected using peak counts, no more individuals were included using the peaks of abundance over the whole ten year period than using two years of abundance data. Nonetheless, to make informed decisions as regards the window of data to be used within a priority site selection exercise requires prior knowledge of long term population trends. Without such knowledge, there can be little or no faith in the resultant protected area networks.

The reason for these patterns most likely relates to the population trends shown for these species over the period 1980/81-1998/99. For many sites, the numbers of wintering waterbirds supported either increased significantly or remained essentially stable (see Chapter 3). Thus, using mean counts, as the numbers of years of abundance data included in the algorithm increased, sites where the trend has been towards an increase in the numbers of birds are less likely to be selected as the mean number of individuals will be less than for those sites supporting more stable populations. In consequence, as the number of years of abundance data included in the selection process increased, the numbers of individuals supported by the resultant hypothetical protected areas networks decreased compared with networks selected using fewer years of data.

Novel networks

For any novel network of protected areas the expectation is that, through its creation, some set of species, assemblages or habitats will be protected from extinction (Pressey et al. 1993, Shafer 1999). Unfortunately, the selection and classification process is often confounded by various political, social and economic constraints, which prohibit the efficient reservation of suitable areas. In consequence, the attainment of overall network efficiency will be secondary to officially awarding protection to biologically important sites. Nonetheless, in the analyses reported here, although there is always an improvement in the numbers of birds in the optimal solutions as the numbers of sites to be selected was increased, a plateau value was attained rapidly (in each case approximately 5% of the sites were required to incorporate 50% of the total number of birds; see Fig. 5.2). Thus, for countries where there are relatively few existing reservations, the initial classification/designation of a small number of carefully selected sites will inevitably contribute significantly more to national conservation targets compared with a random selection of many more sites. Provided of course that the distribution of wintering waterbirds on wetland sites in Great Britain mirrors that of other taxa and other localities.

Established networks

The formal protection of reserve networks has tended to be opportunistic, primarily as a consequence of poor biological data, competition from other valued land use, and the lack of time, resources and personnel (Freitag et al. 1998, Margules et al. 2002, Gaston and Rodrigues 2003). Nevertheless, for certain countries, including the UK, the superior resources available for biodiversity conservation impose fewer constraints on the number of sites potentially available for reservation. Previous analyses have, however, demonstrated the benefits of a re-evaluation of the current SPA selection methodology, and in particular, the adoption of species-specific representation targets as an alternative to the current blanket use of 1% threshold levels for population size (discussed in more detail in Chapters 2-4). In conjunction with this recommendation, from these analyses it is clear that the current use of five-year means is not the most effective method for the selection of priority sites for wintering waterbirds. Indeed, not only was the use of peak counts across years more effective than means overall, where greater numbers of sites were included within the site-selection algorithm, the format of the abundance data incorporated into the reserve selection framework was disproportionately more important. Thus, where there are more sites to be systematically selected, as the case for the UK, using the peaks of abundance will produce more efficient networks as compared with the mean counts.

5.5 Conclusions and implications

The results of this analysis demonstrate the utility of linear programming to address priority site selection for biodiversity conservation. Indeed, today's algorithms, in particular optimisation methods, are able to utilise the available data with the maximum possible efficiency (Pressey and Cowling 2001, Rodrigues and Gaston 2002). Analyses carried out in this way should, therefore, facilitate increased accountability and transparency in the planning process, which is vital should conflicts over individual sites arise. In this respect, I conclude that hypothetical reserve networks selected using peak counts across years, incorporating the maximum numbers of sites were the most effective in capturing wintering waterbird numbers across wetland sites in Great Britain. Further, changes to the current means by which SPAs are classified in the UK might usefully be recommended, to take account of the more effective methodology shown by these analyses.

CHAPTER 6:

EXPLORING FLEXIBILITY AND IRREPLACEABILITY

6.1 Introduction

The identification and management of priority areas has long been recognised and accepted by conservation planners as the most practical option for biodiversity conservation (Soulé 1991). Indeed, an ever increasing human population coupled with already high population densities, progressive land use change, political instability, war, limited/inadequate funds available for conservation purposes and competition from other legitimate development options inevitably impose heavy restrictions on conservation activities. Despite our best efforts we cannot protect all elements of biodiversity in all possible locations (Pressey and Taffs 2001, Margules et al. 2002). Unfortunately, how best to prioritise areas for conservation is not straightforward, and has been the subject of intense and often circular debate for several decades (see Kingsland 2002 for a review). Nonetheless, as the true extent of the biodiversity crisis is realised and translated into potential impacts on society, the selection of priority areas for conservation has received increasingly focussed attention (Pressey et al. 1993, Margules et al. 2002, Salafsky et al. 2002).

Historically, protected areas have tended to be selected on an *ad hoc* or opportunistic basis according to availability, lack of competition with alternative development proposals, or scenic value (Pressey et al. 1994; see also Chapter 5). However, more recently (in the last 15-20 years) various more systematic methods have been proposed and tested for the problem of identifying a representative network of protected areas. These have progressed from simple scoring or ranking methods (e.g. Margules and Usher 1981, Smith and Theberge 1986, Lee et al. 2001), to iterative heuristic approaches (e.g. Kirkpatrick 1983, Margules and Nicholls 1987, Pressey and Nicholls 1989, Vane-Wright et al. 1991, Rebelo and Siegfried 1992a, Ryti 1992, Underhill 1994, Williams and Revelle 1996, Freitag et al. 1997, Williams and ReVelle 1997, Nantel et al. 1998, Clemens et al. 1999, Pressey et al. 1999, Curio 2002, Justus and Sarkar 2002, McDonnell et al. 2002, Nalle et al. 2002, Rosing et al. 2002), and optimisation algorithms (e.g. Chapters 4, 5 and 7; Davis et al. 1999, Rodrigues et al. 1999, 2000b, 2000c, Polasky et al. 2001, Rodrigues and Gaston 2001, 2002). Simple scoring methods were summarily dismissed by Pressey and Nicholls (1989a) for their inefficient representation of required features. Similarly, iterative heuristic methods, which are still championed by many, have also been strongly criticised as they cannot guarantee optimality, nor can they determine how far from optimality solutions actually are (Rodrigues and Gaston 2002). It is for this reason that optimisation methods, techniques from the field of operations research, are gradually becoming accepted as the most suitable and indeed efficient means of identifying networks of priority sites for biodiversity conservation, particularly given the advances in computer processor power in recent years. Despite differences in formulation and performance, central to each of these priority site selection procedures is the aim to identify key areas for conservation in the most efficient, effective and accountable manner (Margules et al. 2002). Fundamentally, the intention is to develop a representative system of reserves, which can be argued with as much conviction as any legitimate competing land use proposal. In this respect, Pressey et al. (1993) put forward a set of three objectives to be considered throughout the priority site selection process, namely: complementarity, flexibility and irreplaceability.

Complementarity refers to the degree to which component sites of a protected area network contribute new features to the network rather than simply providing duplicates of those already protected (see Vane-Wright et al. 1991, Pressey et al. 1993, Cabeza and Moilanen 2001, Margules et al. 2002 for reviews). This principle requires the selection of networks of priority sites rather than individual sites in isolation, and is closely related to the notion of network efficiency (*sensu* (Pressey et al. 1993; discussed in detail in Chapter 4). Essentially, minimising redundant duplication (beyond any duplication that is considered desirable) and ensuring that component sites are complementary in the features they contain will maximise network efficiency (Pressey et al. 1993). It is not, therefore, inevitable that complementary networks contain the richest areas (Knopf and Sampson 1994).

Flexibility is a property of networks of protected areas and relates to the numbers of alternative network configurations possible that solve a particular reserve selection problem. For any given region there are likely to be many different ways to satisfy the constraints posed by a priority site selection algorithm. In their basic form, however, optimisation algorithms are limited in that they do not address flexibility and provide no information as to the priority value of individual selected sites (Ferrier et al. 2000). Rather, these approaches identify a single network of sites to satisfy conservation targets in the most efficient manner. Single runs of an algorithm, whilst having the benefit of rapid acquisition, do not provide planners with all the information necessary to make an informed decision regarding the location and configuration of a reserve network and to adapt to any changes in site availability (Arthur et al. 1997), unless of course there is only one optimal solution (see Chapter 7). Nonetheless, simple modifications to the basic site selection algorithm enable all possible optimal solutions to be found.

The value of exploring flexibility lies with negotiation power, which makes it one of the most important attributes of a systematic reserve selection process (Nicholls and Margules 1993, Pressey et al. 1993, Williams 1998, Rodrigues et al. 2000a, Margules et al. 2002). Indeed, because, in most cases, there will be alternative development proposals for priority conservation areas (Prendergast et al. 1999, Possingham et al. 2000), planners are forced to put forward their case and to negotiate at an equivalent level to other interested parties.

There are several different definitions of irreplaceability (Ferrier et al. 2000), principally it refers to the contribution each potential protected area makes to achieving a particular conservation target (Pressey et al. 1994). Amongst all possible optimal solutions to a particular reserve selection problem, irreplaceability refers to the likelihood that a given site will need to be protected to satisfy a specific set of conservation targets (Ferrier et al. 2000) and provides an indication of where and when to allocate conservation resources (Pressey and Taffs 2001). This measure provides an indication of the relative importance of individual sites for conservation and of the order in which sites should be acquired or protected.

A site may be considered irreplaceable for three reasons: first, because it contains one or more unique features; second, because it contains one or more non-unique features and the conservation goal is equal to their total remaining extent; or third, because the site contains occurrences of one or more non-unique features that are sufficiently large that the goal cannot be achieved without conserving that area (Margules et al. 2002). All replaceable areas are negotiable, whereas irreplaceable areas are not. Fundamentally, without irreplaceable areas representation targets cannot be achieved (Pressey et al. 1993, Ferrier et al. 2000). Conversely, areas with lower irreplaceability have progressively more replacements, less likelihood of being required to satisfy conservation targets, and a lower impact on overall biodiversity should the site be destroyed or rendered unavailable (Ferrier et al. 2000).

Accepting the importance of these three principles (complementarity, flexibility and irreplaceability) for conservation planning, using wetland sites in Great Britain as a case study, this chapter explores the flexibility and irreplaceability of optimal networks of sites selected using a linear programming optimisation algorithm. In addition, the tradeoffs between the level of species representation, the number of sites in the optimal solution, and solution flexibility are examined.

6.2. Methods

Data

Analyses were conducted for 17 species of migratory waterbird using data from WeBS. Data for the ten year period, 1989/90 to 1998/99 inclusive, were included for Bewick's swan, whooper swan, European white-fronted goose, dark-bellied brent goose, shelduck, wigeon, gadwall, teal, mallard, pintail, shoveler, pochard, tufted duck, goldeneye, smew, red-breasted merganser and goosander. For these analyses 1962 sites were included, the same set for each of the 17 species and in all years.

Site totals

The total bird count at each individual wetland site was calculated for each of the 17 species over the ten year period 1989/90-1998/99. In line with the methodology adopted by WeBS (see Pollitt et al. 2000), these site counts were taken as the maximum of the individual monthly counts from September to March. For each species individually, any site for which no individuals were recorded over the ten year period of this analysis were excluded from the analysis.

Network selection algorithms

The site-selection approach used in this paper is a modified minimum area problem (Pressey et al. 1997; Chapter 4), based on the principle of complementarity. Originally developed for operations research, these mathematical reserve selection methods aim to maximise the representation of natural features (e.g. species or habitats) in the smallest possible area or at the lowest cost (Pressey et al. 1997). However, rather than simply attempting to represent all species a given number of times as has been the tradition for these types of analysis (e.g. McKenzie et al. 1989, Sætersdal et al. 1993, Pressey et al. 1994, Price et al. 1995, Williams and Revelle 1996, Csuti et al. 1997, Polasky et al. 2000, Rodrigues et al. 2000b), the basic algorithm used here selects from all possible sites (in this case N = 1962) the network that minimises the total number of sites selected whilst satisfying numerical representation targets for each of the 17 waterbird species individually.

Representation targets for each of the 17 species were set from ≥10% of the national total, taken as the sum of the 1962 individual site totals for that species, to ≥90% of the national total at intervals of 10%. In each case, if the actual representation target was not feasible, the species representation level was set to the maximum possible. In actual fact, these targets only required modification for certain species at the ≥80% and ≥90% level.

The peaks of abundance for each of the 17 species individually over the ten year period 1989/90-1998/99 were used within the selection algorithm as this was identified as the most efficient and effective means of selecting priority sites for these species (Chapter 5). Throughout, optimal solutions to all algorithms were determined using CPLEX™ linear programming software (ILOG 1999).

Flexibility

Flexibility was assessed using the method outlined by Arthur et al. (1997). After finding the initial optimal solution (minimum set), this combination of sites was added to the minimum area algorithm as a further constraint. This forces the algorithm to select novel combinations of sites within the confines of the algorithm constraints. Prior to each run of the algorithm, the sites-species matrix was randomly re-ordered to ensure that the solutions were not sequential as is normally the case using CPLEX™ optimisation software. This procedure is important where it is not deemed sensible to find all optimal solutions (i.e. where the data matrix is particularly large). In such cases, a random subset of 500 solutions was found. For each representation level, this process was repeated until the number of sites increased from the minimum set (i.e. the solution was no longer optimal). Flexibility was measured both as the number of minimum sets found for each representation level and also as the total numbers of sites selected at least once amongst these same minimum sets.

Irreplaceability

The irreplaceability value (*I*) of a site refers to the frequency of selection amongst all minimum sets for each level of representation (10-90%), expressed as a percentage of the number of possible occurrences (i.e. the total number of minimum sets found). The total and mean (±SD) *I* value for each minimum set was calculated as the sum of the *I* values generated for each site included in that subset.

6.3 Results

Minimum sets

As the level of representation increased from 10% to 90% of the national total, the optimal numbers of sites required increased from a subset of nine sites for 10% to 162 for 90% (Table 6.1; Fig. 6.1; Figs 6.4-6.12). These subsets, however, correspond to only 4.5% of the total number of wetland sites included in the algorithm (*N* = 1962) for 10% representation, to 8.3% for 90% representation. Thus, to represent all 17 species at the 90% level (or the maximum possible) required less than 10% of the total numbers of wetland sites. At the 10% representation level, the looping constraint was stopped after 500 additional constraints were added to the algorithm (i.e. 500 optimal solutions were found). Thus, the actual number of optimal solutions for a representation target of 10% is unknown, but could easily be determined over a longer time period. However, 500 possible solutions is more than enough to provide planners with sufficient flexibility to construct solid and defensible arguments for priority site selection.

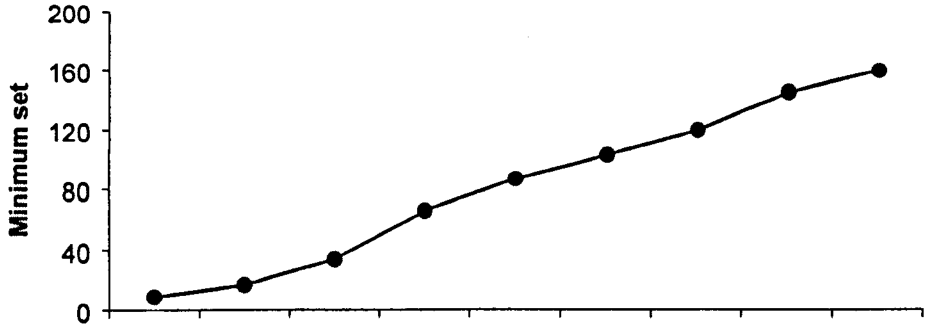

Figure 6.1: Tradeoffs between the representation level achieved (10-90%) and the size (numbers of sites) of the minimum set.

Flexibility

The numbers of minimum sets (optimal solutions) found, as a function of the representation level is not straightforward. While generally decreasing as the representation level increases from 10% to 90%, the single solution found for 20% representation was unexpected (Table 6.1). Nonetheless, the generally lower number of optimal solutions for higher levels of representation suggests that there is a tradeoff between the level of species representation achieved and the flexibility of the minimum set (Table 6.1 and Fig. 6.2).

Table 6.1 For each representation target (10-90%), the size of minimum set (number of sites), the number of possible combinations of *N* sites (where *N* = minimum set size), the number of minimum sets, the total number of sites selected at least once, the mean I value and standard deviation (SD) for each individual site across all minimum sets and the number of irreplaceable sites.

Target	Minimum set	Possible combinations	Number of minimum sets	Total sites selected	Mean I	SD	Irreplaceable sites
10	9	4.23E+29	500+	94	9.68	24.16	3
20	17	2.48E+41	1	17	100.00	0	17
30	34	2.27E+73	19	46	73.91	38.99	26
40	66	1.26E+124	17	91	72.53	40.27	52
50	88	4.26E+154	7	104	84.62	31.08	77
60	105	2.94E+176	3	117	89.74	22.94	96
70	122	1.11E+197	2	127	96.06	13.52	117
80	148	2.76E+226	2	159	93.08	17.32	137
90	162	2.28E+241	1	162	100.00	0	162

Overall, as the level of representation increased, flexibility decreased, although the correlation was not statistically significant ($r = -0.566$; $N = 9$; $P > 0.05$; Fig. 6.2). However, if 20% representation is omitted, this correlation becomes statistically significant ($r = -0.915$; $N = 8$; $P < 0.01$).

The total numbers of sites selected at least once amongst all minimum sets for each representation level increased from 94 for 10% to 162 for 90% representation (Table 6.1; Figs 6.4-6.12). However, the numbers of sites selected to satisfy the 20%, 30% and 40% representation targets were less than the 94 sites selected for the 10% target (Table 6.1).

Irreplaceability

The mean *I* value (average of the total irreplaceability values across all minimum sets) of the minimum sets for each representation level generally increased as the representation level increased (Table 6.1 and Fig. 6.3). This was expected given that the numbers of irreplaceable sites (i.e. those present in all optimal solutions) also increased as the representation target increased (Table 6.1).

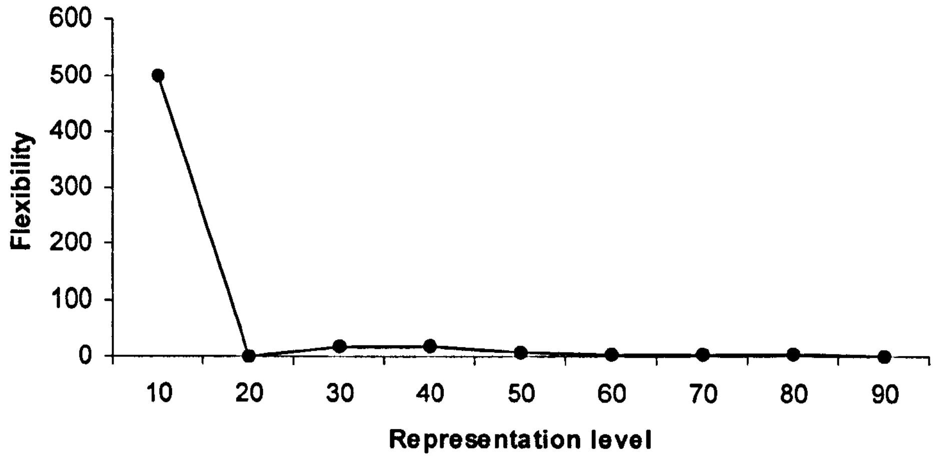

Figure 6.2: Tradeoff between the representation levels achieved for each of the 17 species and the minimum set flexibility for 10-90% representation.

Given that the 20% representation target only generated a single optimal solution, each selected site is inevitably irreplaceable in this case. This accounts for the departure from the overall trend for 20% representation (Fig. 6.3). Figs 6.4 to 6.12 show the spatial location and *I* value for each site selected at least once for each of the representation levels.

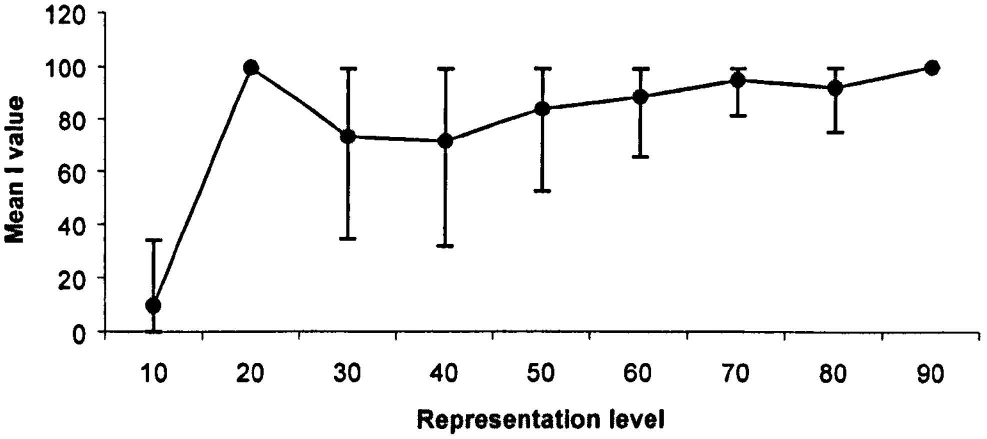

Figure 6.3: Tradeoffs between the level of representation achieved for each of the 17 species and the mean I value (±SD) for sites appearing in at least one minimum set.

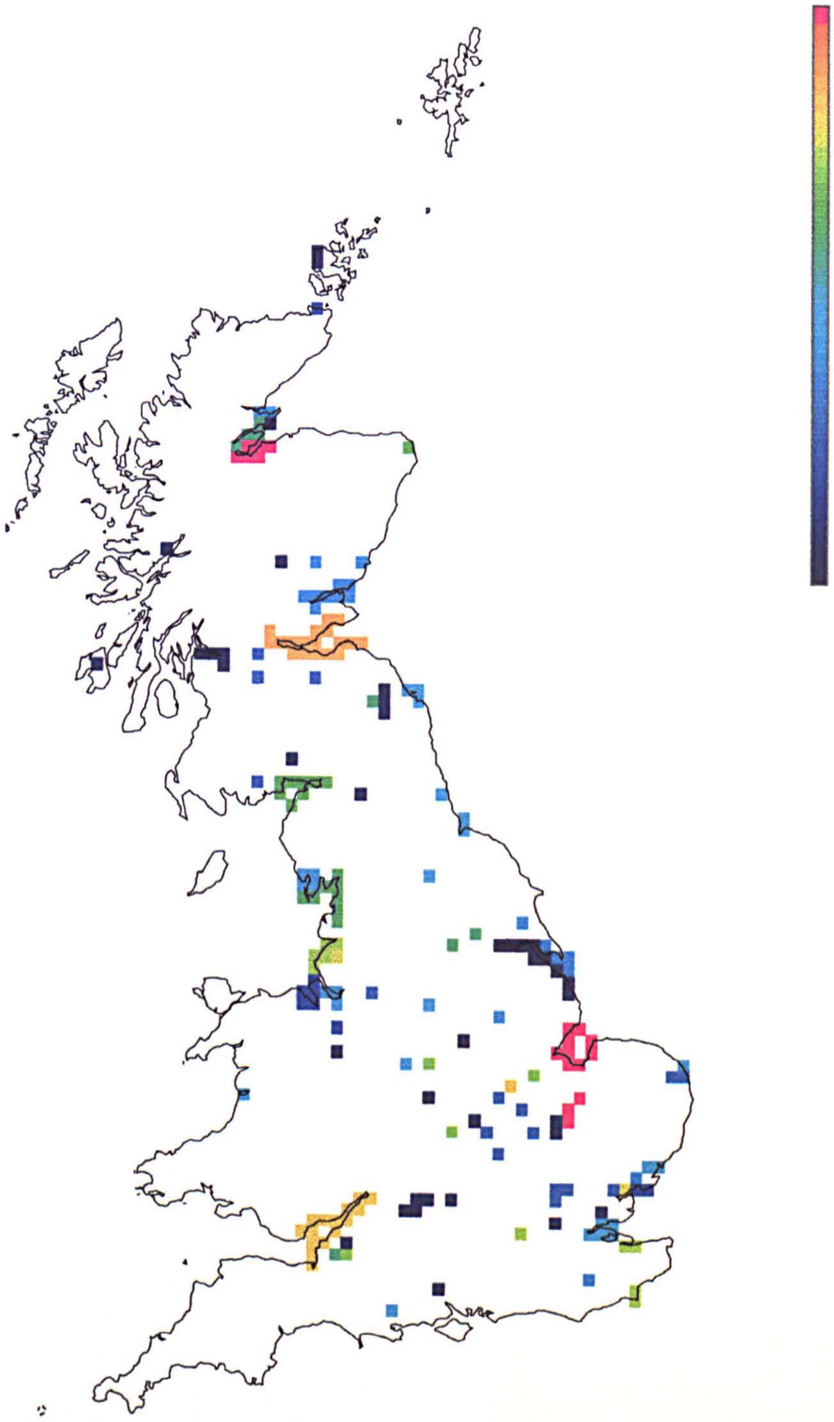

Figure 6.4 Location of all sites selected at least once using the 10% representation target. The equal interval scale represents a decrease in irreplaceability and runs from red (appears in all alternative networks) to navy blue (appears in a single network). NB. Individual sites may comprise more than one 10km square.

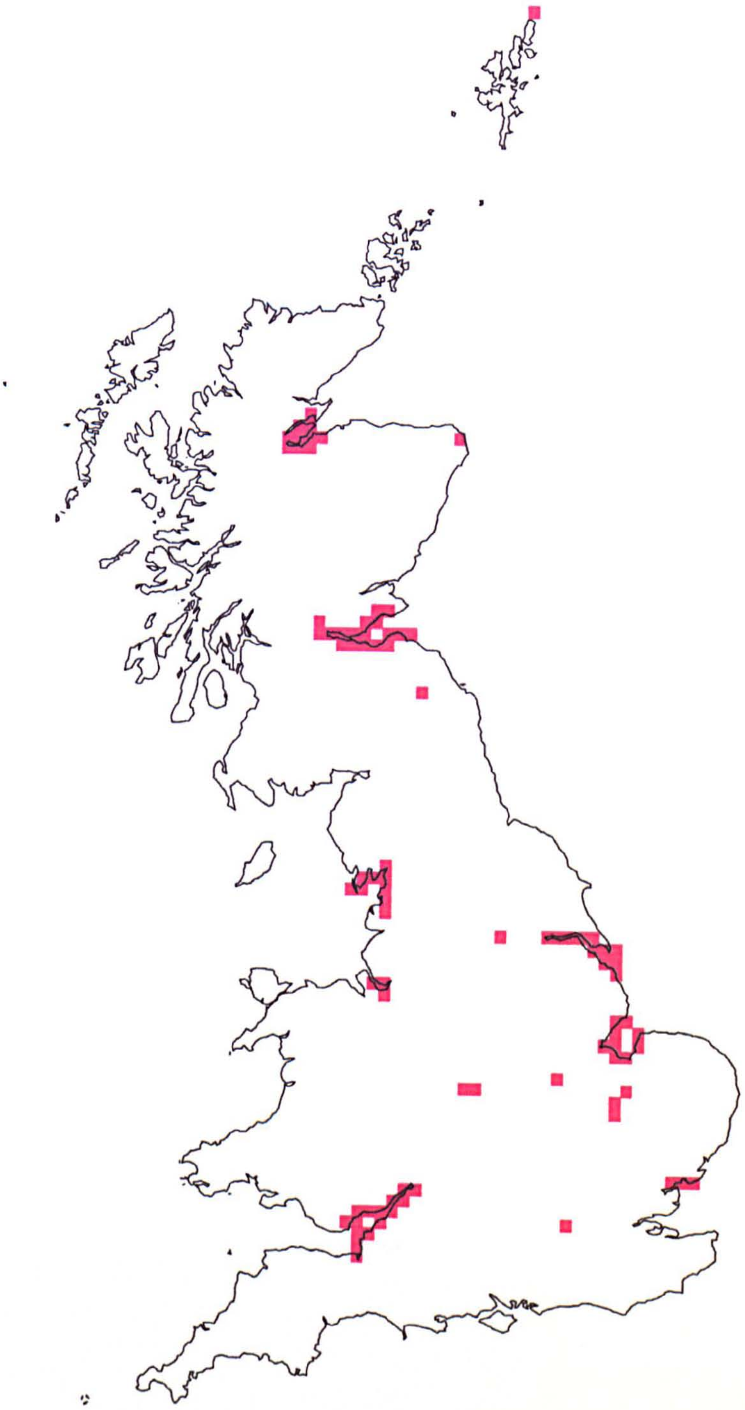

Figure 6.5: Location of all sites selected using the 20% representation level (red squares). As only one solution was found all sites are irreplaceable.

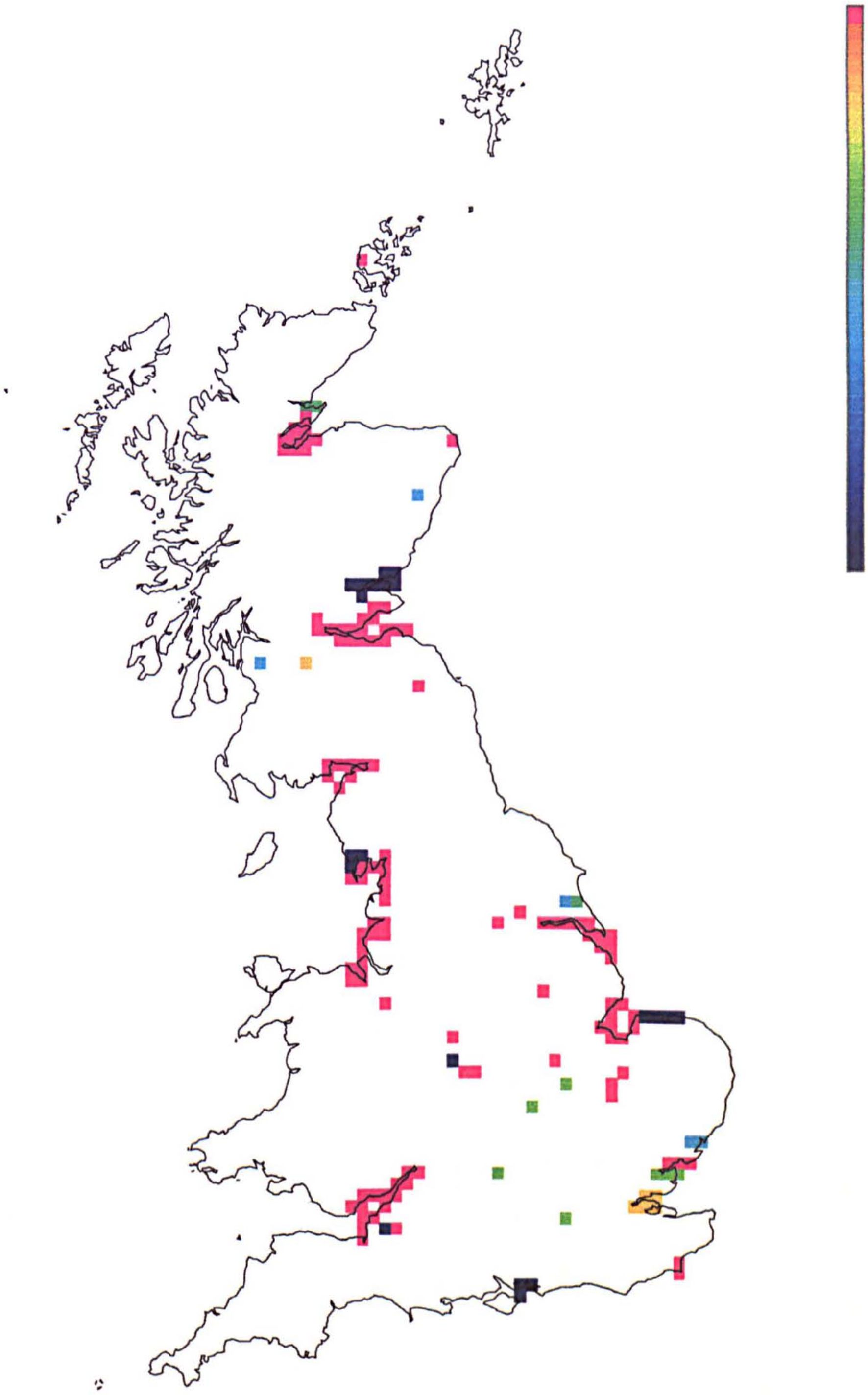

Figure 6.6: Location of all sites selected at least once using the 30% representation target. The equal interval scale represents a decrease in irreplaceability and runs from red (appears in all alternative networks) to navy blue (appears in a single network). NB. Individual sites may comprise more than one 10km square.

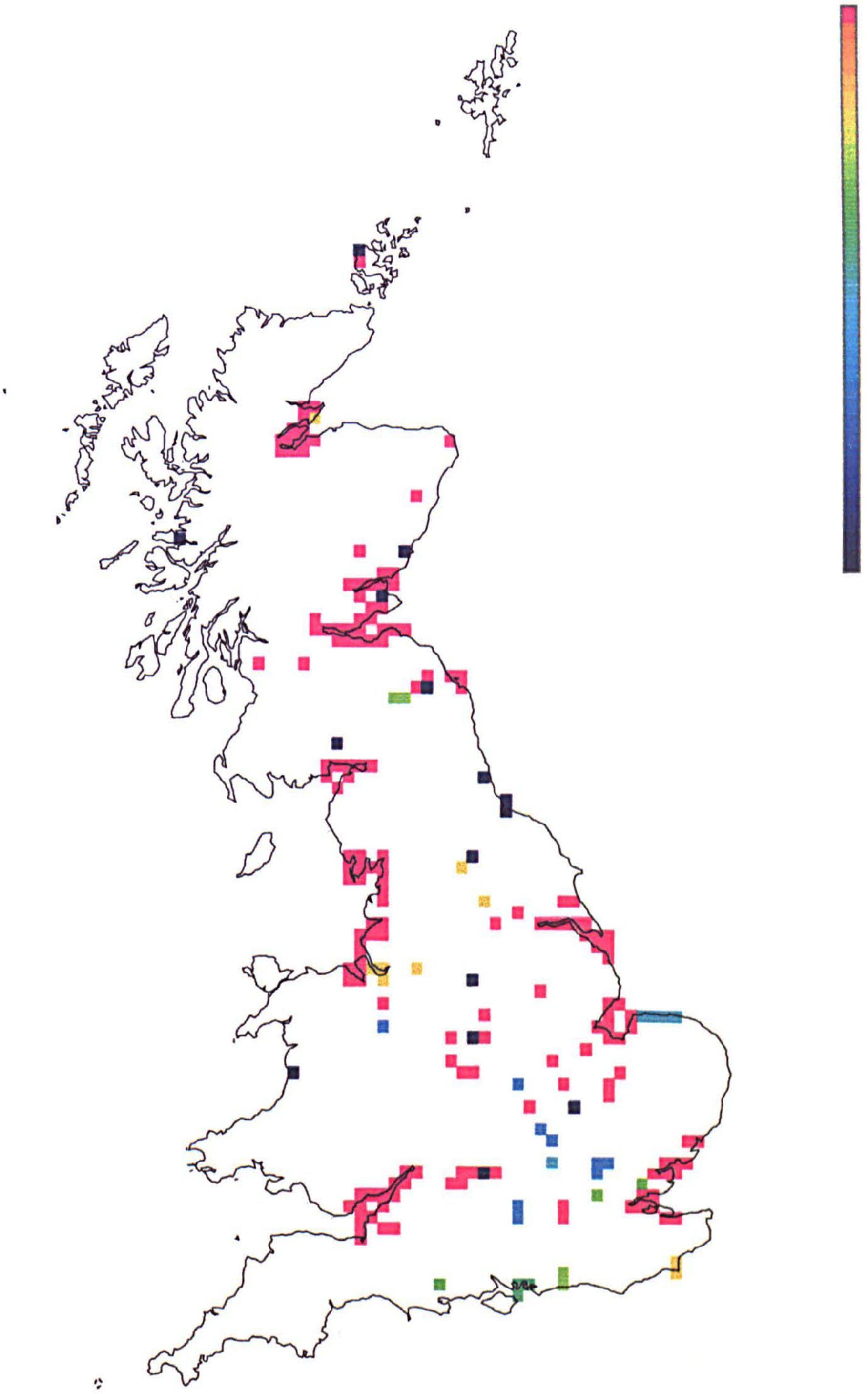

Figure 6.7: Location of all sites selected at least once using the 40% representation target. The equal interval scale represents a decrease in irreplaceability and runs from red (appears in all alternative networks) to navy blue (appears in a single network). NB. Individual sites may comprise more than one 10km square.

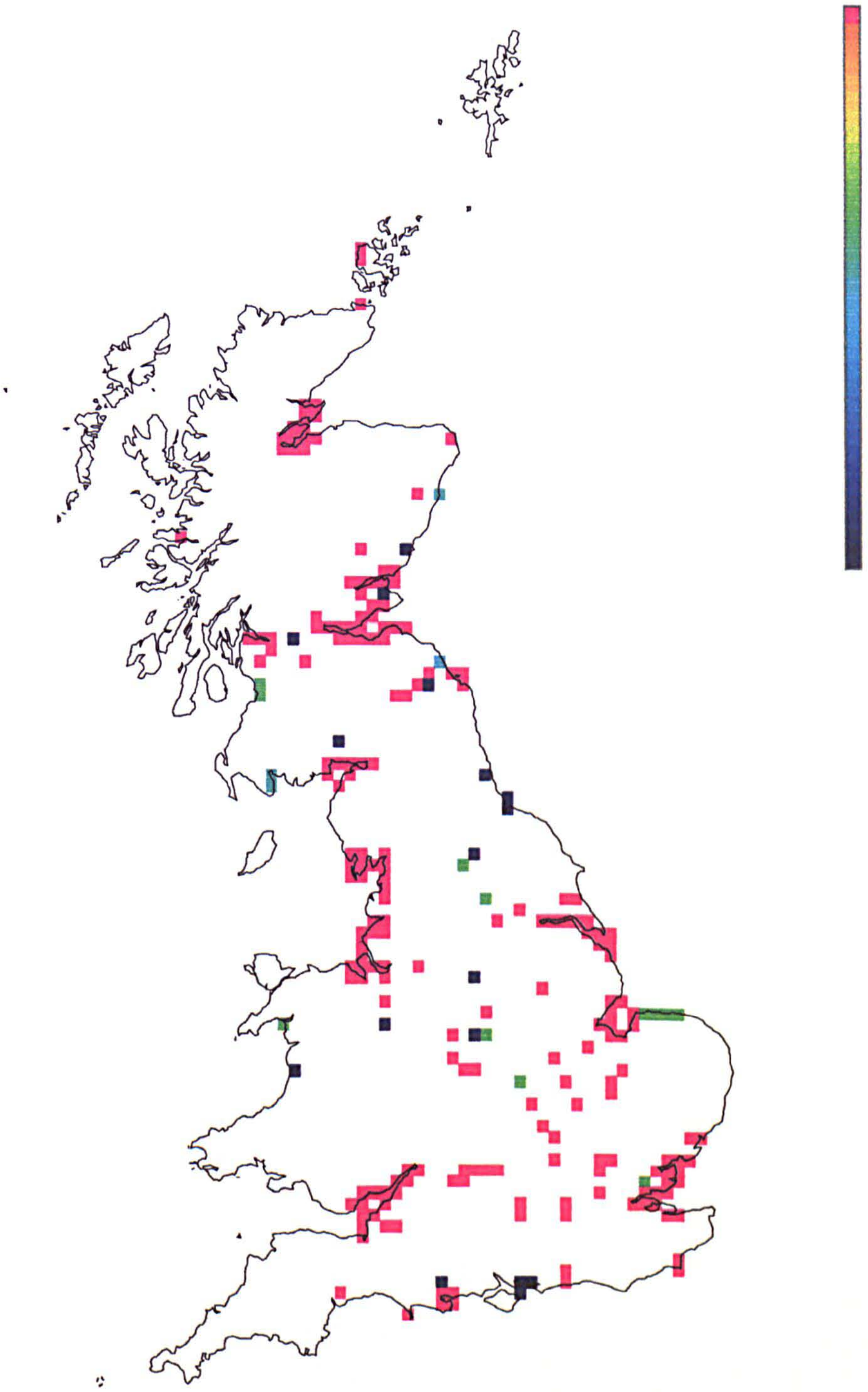

Figure 6.8: Location of all sites selected at least once using the 50% representation target. The equal interval scale represents a decrease in irreplaceability and runs from red (appears in all alternative networks) to navy blue (appears in a single network). NB. Individual sites may comprise more than one 10km square.

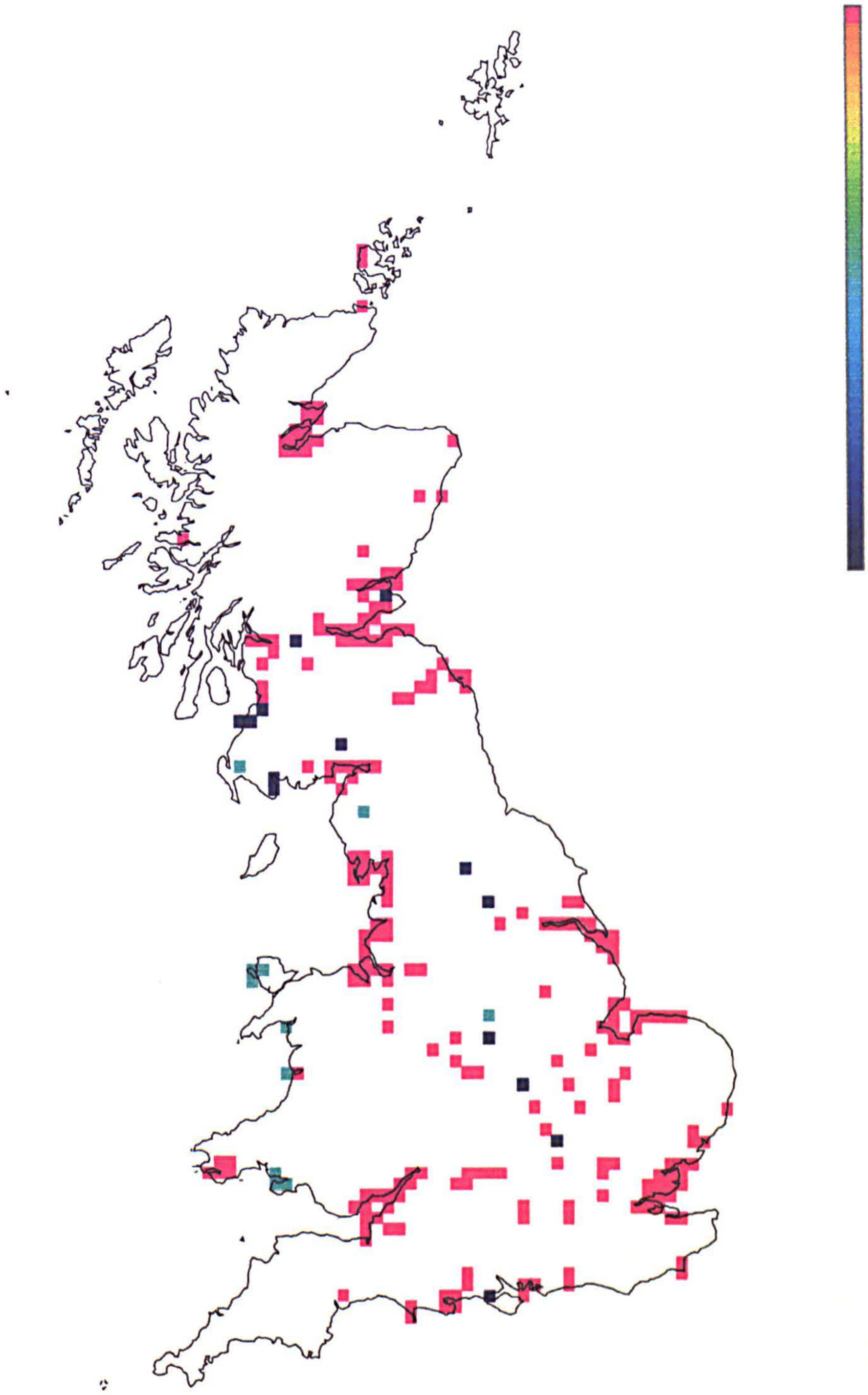

Figure 6.9: Location of all sites selected at least once using the 60% representation target. The equal interval scale represents a decrease in irreplaceability and runs from red (appears in all alternative networks) to navy blue (appears in a single network). NB. Individual sites may comprise more than one 10km square.

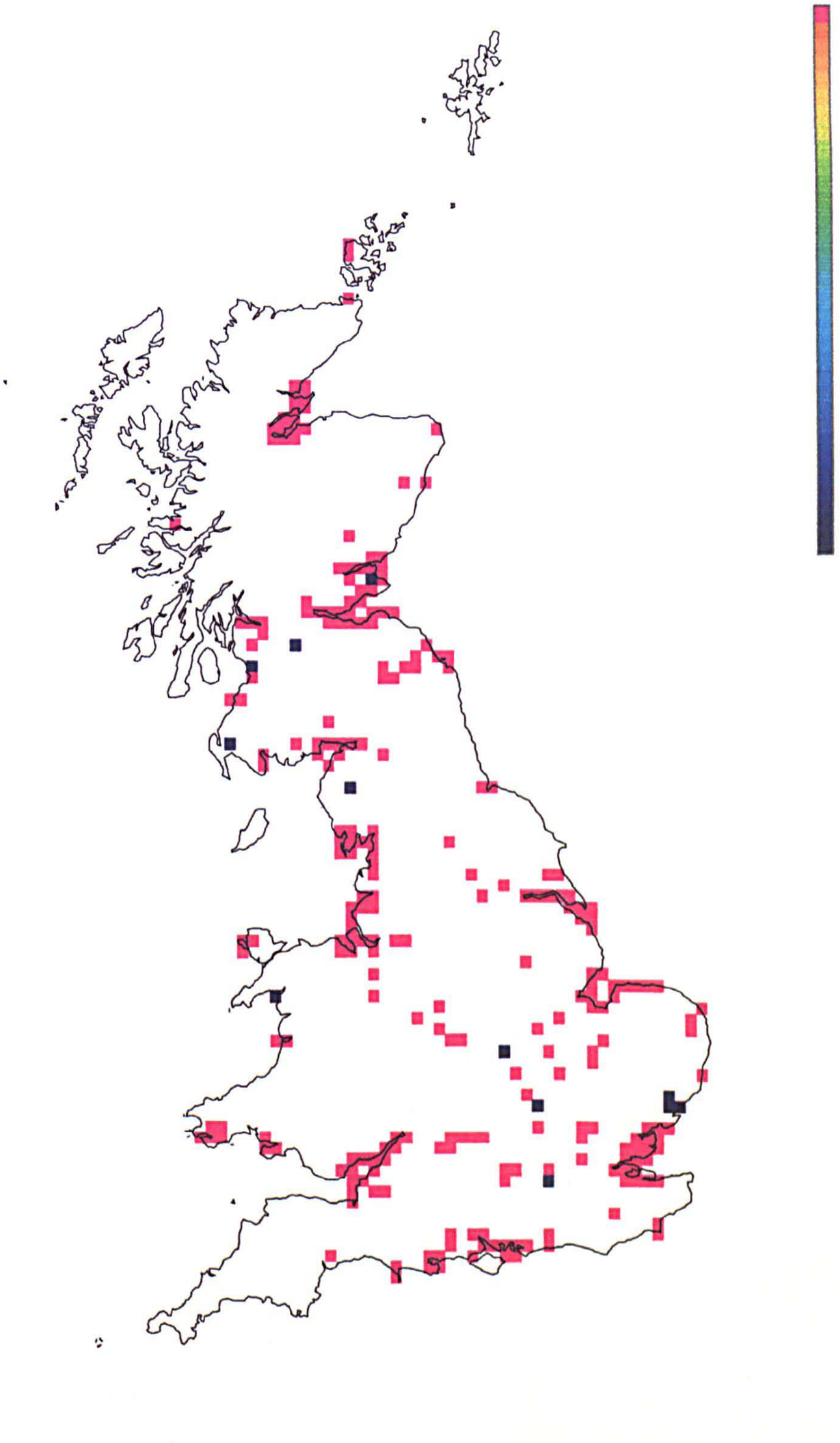

Figure 6.10: Location of all sites selected at least once using the 70% representation target. The equal interval scale represents a decrease in irreplaceability and runs from red (appears in all alternative networks) to navy blue (appears in a single network). NB. Individual sites may comprise more than one 10km square.

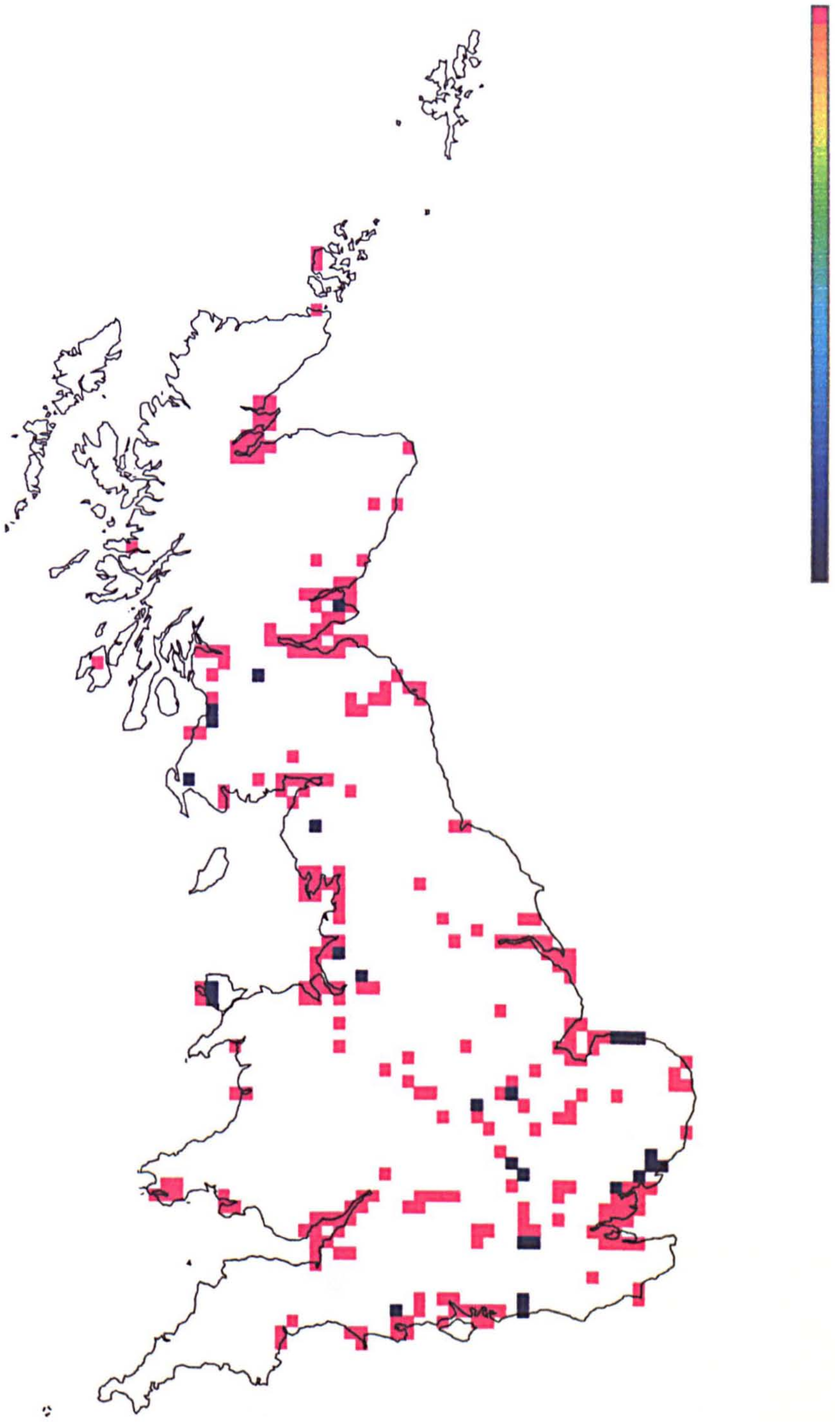

Figure 6.11: Location of all sites selected at least once using the 80% representation target. The equal interval scale represents a decrease in irreplaceability and runs from red (appears in all alternative networks) to navy blue (appears in a single network). NB. Individual sites may comprise more than one 10km square.

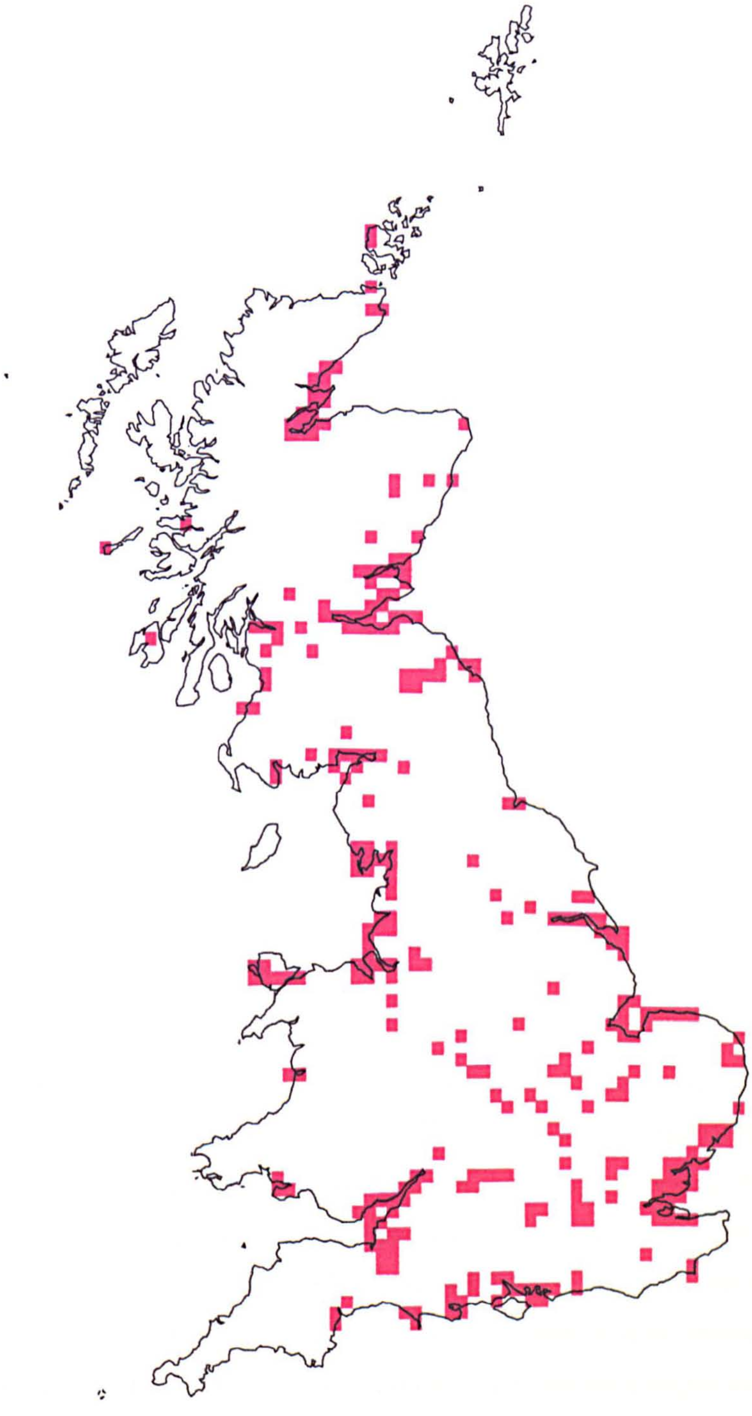

Figure 6.12: Location of all sites selected using the 90% representation level (red squares). As only one solution was found all sites are irreplaceable.

For each representation target, the minimum set with the highest overall *I* value (sum of individual site scores) is considered to be the top priority network. There is one such networks for 10%, one for 30%, one for 40%, four for 50%, one for 60%, two for 70%, two for 80% and one for 90% representation. Inevitably, given that there was a single minimum set found using 20% and 90% of the national total for each species, there is only one such top priority network for these levels of representation (Fig 6.3).

Table 6.2: For each representation level (10-90%), the maximum *I* value (relating to the minimum set with the largest total irreplaceability) calculated as the sum of the individual site *I* values, the mean *I* value across all minimum sets and the standard deviation in these values. An *I* value of 100 means that all sites are irreplaceable as there is only one single minimum set possible (i.e. for 20% and 90%).

Representation level	Maximum *I* value	Mean *I*	SD
10	688.90	76.54	29.27
20	100.00	100.00	0
30	3147.38	92.57	20.45
40	6370.61	96.52	10.11
50	8542.83	97.08	10.65
60	10100.10	96.19	13.33
70	11950.00	97.95	9.95
80	14150.00	96.26	13.20
90	100.00	100.00	0

In general, the mean *I* value for these top networks increased as the representation increased (Table 6.2). Additionally, the number of irreplaceable sites also increased from three for 10% representation to 137 for 80% (Table 6.2). Of note is that the three sites that are 100% irreplaceable for the 10% representation level are also necessary for all other networks (Fig. 6.13). As expected, the greatest numbers of wetland sites were shared between networks selected for 80% and 90% representation (138 sites). Additionally, 121 sites were shared between networks selected using 70% and either 80% or 90% representations. Conversely, <10 sites were shared between networks selected using the 20% target and any other network.

Considering all minimum sets selected across all nine levels of representation only 202 sites (10.3% of all sites) were selected at least once (Table 6.1; Fig. 6.14). Additionally, of these 202 sites <7% were irreplaceable

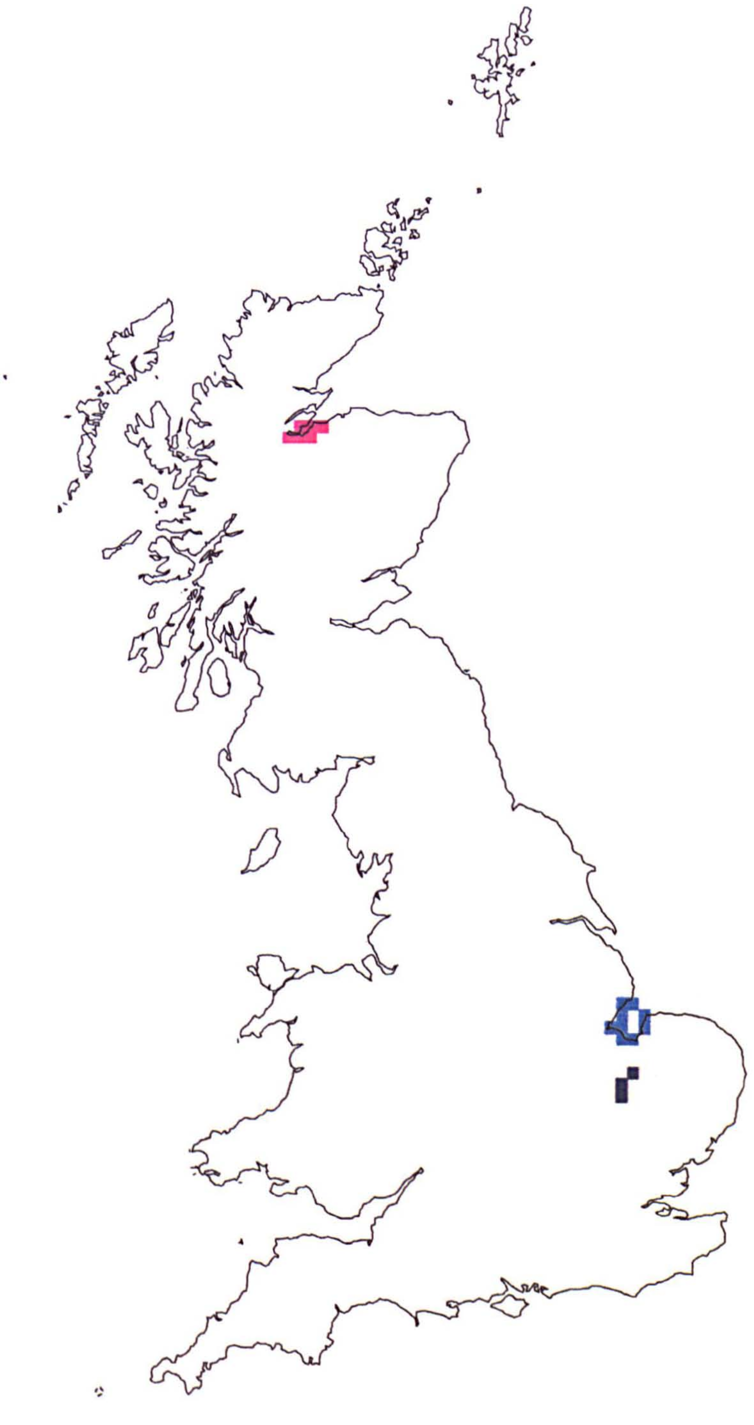

Figure 6.13: Location of the three core sites. These sites are those required for all networks. Red = The Inner Moray Firth, pale blue = The Wash, Navy blue = The Ouse Washes. NB. Individual wetland sites comprise more than one 10km square.

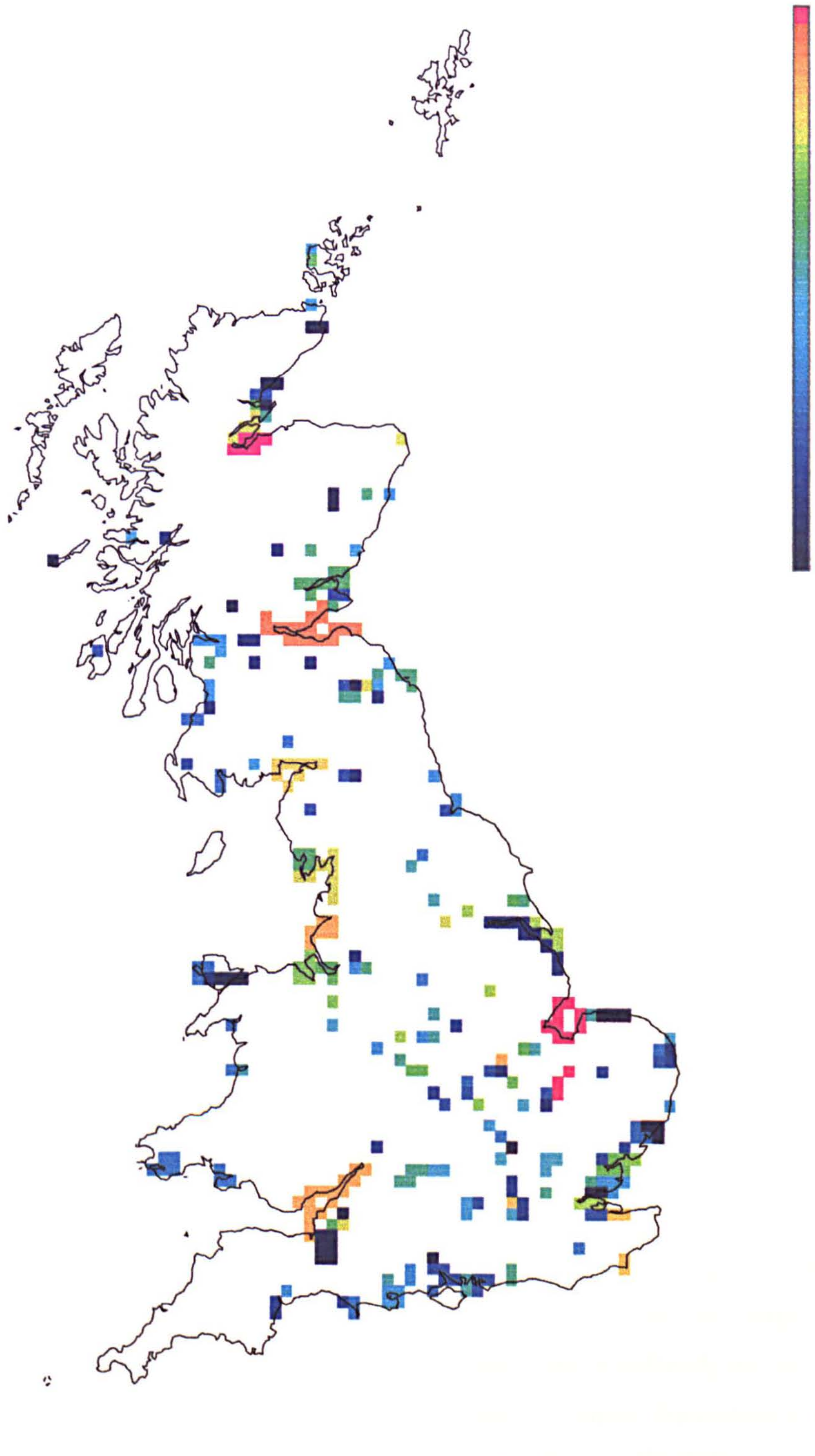

Figure 6.14: Location of all sites selected at least once using all representation levels (i.e. the results of all selection algorithms). The equal interval scale represents a decrease in irreplaceability and runs from red (appears in all alternative networks) to navy blue (appears in a single network). NB. Individual sites may comprise more than one 10km square.

6.4 Discussion

The strength of the systematic site selection procedure adopted in this chapter lies in its efficiency, effectiveness, accountability and flexibility. Further, this technique is powerful, yet conceptually straightforward. Thus, apart from being advantageous to the user, this means the process and more importantly the results themselves can be easily explained to and repeated by interested parties. Additionally, the process itself forces planners to be explicit in the targets and objectives for a particular conservation network, and enables the exploration of flexibility.

Ferrier et al. (2000) and Pressey and Taffs (2001) suggested that the exploration of flexibility and, therefore, irreplaceability is not a feasible objective when dealing with large data sets because of the sheer numbers of possible optimal solutions. Indeed, the alternative sets of sites that can achieve a particular conservation goal may be very numerous (Rebelo and Siegfried 1992b, Pressey et al. 1994, Ferrier et al. 2000). However, all sites are not equal in terms of their size, quality and numbers of both species and individuals, therefore, a large proportion of such combinations will not satisfy the constraints posed by the selection process. Inevitably, therefore, the numbers of optimal solutions will always be less, and usually substantially less, than the number of possible combinations. If the number of optimal solutions corresponded exactly to the number of possible combinations, rather than using site selection algorithms it would be as effective simply to select a given number of sites at random to achieve the conservation targets. Moreover, and contrary to Ferrier et al. (2000) and Pressey and Taffs (2001), the linear programming software used throughout this thesis (i.e. Chapters 4-7) has the advantage that solutions can be obtained in fractions of a second. For example, the average (±SD) processing time to select networks of sites to satisfy the 10% target was 1.78 (±0.76) seconds, with 431.10 (±195.71) iterations. Similarly, to identify networks for the 50% target took just 0.09 (±0.03) seconds with 45.80 (±4.87) iterations, whereas networks for the 70% target took 0.08 (±0.02) seconds and 41.50 (±12.02) iterations. This means that a thorough exploration of flexibility takes only a matter of minutes in most cases, (e.g. 7.58 minutes in total for 30%, 1.60 minutes for 40% and 0.47 minutes for 50% representation), which is a small price to pay for increasing the validity and utility of the process for real world conservation planning.

Inevitably, certain sites will not be available for conservation purposes, given the realities of the competing demands of society. However, fully exploring the options for optimal protected area networks will inevitably dramatically improve the planning and negotiation phase. The single exception to this is the 10% representation network, where exploration was terminated after 500 solutions were found. Nonetheless, rather than being a time constraint, such a number was considered sufficient to allow for adequate flexibility during any planning process. In such circumstances, the numbers of solutions produced will depend on the degree to which possible networks conflict with alternative development proposals.

Minimum sets

The advantage of minimum sets is manifest where conservation agencies are constrained in the numbers of sites they can buy or manage (Woinarski et al. 1996). Indeed, ensuring an efficient network of reserves enables planners to make the best use of scarce resources. Inevitably, however, as the level of representation for each species was increased from 10% to 90% of the national total, there was a corresponding increase in the size of the minimum set, indicating a tradeoff between the level of representation achieved and the likely cost of the reserve network. This increased cost assumes that more sites will equate to higher acquisition costs (including legal expenses, expert advice and land acquisition).

Determining the best size of a protected area network is no simple task. Indeed, this process is complicated/confounded by the availability of resources, desired targets for species representation, and issues of population viability. In particular, for the majority of countries there will be a fixed and generally small budget allocated to site purchase. Furthermore, there is considerable variation in political, economic and social conditions between countries (O'Connor et al. 2003) and competition for land, particularly in developed countries, is fierce (Seymour et al. 2001). This means that only a certain number of sites can realistically be suitably protected. Thus, this financial restriction will, ultimately, dictate the maximum possible level of representation across the network as a whole. For example, if the numbers of sites were restricted to 34, then in the present case, and other considerations aside, the maximum possible level of representation would be 30% of the national total for each species.

Equally, setting appropriate/desired conservation targets will dictate the size of the minimum set and therefore a substantial component of the overall cost of the project. For example, for the individual species included in this chapter, setting the target to 10% of the national total for all target species will incur the costs of nine protected areas (e.g. acquisition, management, monitoring costs). Similarly, a target of 80% for each species will incur the costs for the purchase of 148 sites. This difference in the size of the minimum set likely equates to a substantial difference in expenditure, which may prohibit effective conservation efforts in many countries.

Target setting is not, however, a simple task. Not only do these targets need to be biologically derived and objective (Chapter 4), but also they must aim to take account of species persistence. In this respect, various authors have addressed the concept of species viability within networks of reserves at a local scale (Bedward et al. 1992, Kiester et al. 1996, Pressey et al. 1996, Nicholls 1998, Rodrigues et al. 1999), and at a regional (within reserve network) level (Margules et al. 1994, Williams and Araújo 2000). To take account of species persistence means that there will be a lower limit on the target set in an attempt to ensure population viability. The cost in terms of representation should be offset by the benefits of developing a reserve system in which natural pattern and processes are likely to persist in the face of change, and which will be implemented so that threatening processes have a minimal impact on conservation targets (Cowling 1999). Inevitably, targets set to below a sustainable level will not allow a species to persist in the network regardless of protected status.

Flexibility

Flexibility is essential to promote informed negotiations over possible protected areas with other interested parties and landowners, to encourage compromise with other developments, and to allow conservation objectives to be dealt with and viewed as comparable with other landuse proposals. However, these analyses clearly demonstrate the tradeoff between flexibility and representation. Indeed, as the level of representation is increased, optimal network flexibility decreased. Thus, the higher the conservation target, the fewer options there are for conservation. Consequently, at higher levels of representation planners will be under increased pressure to secure priority sites, as there are few alternatives. This could mean paying over the odds for particular sites where there are competing land use options. In such a situation, these contentious sites would most likely be substituted for other sites wherever possible.

Irreplaceability

The irreplaceability of a site provides an indication of its priority for protection. However, as the level of representation increased, irreplaceability scores for each selected site concurrently increased; this is inevitable given the tradeoff between representation and flexibility. Rebelo (1994) put forward the term 'conditional irreplaceability', referring to those sites which, given a limit to the numbers of sites, are 100% irreplaceable. These sites can, however, be substituted for others if a larger number of sites/area/budget is available. In this respect, depending on the particular level of representation, the identity and number of irreplaceable sites changes. For example, all 162 of the sites for 90% representation are 100% irreplaceable, however only 121 of these same sites are irreplaceable if the target for representation is set to 80%, 105 for 70%, 81 for 60%, 54 for 40%, 27 for 30%, nine for 20% and three for 10%. Rebelo contrasted conditional irreplaceability with 'global irreplaceability', which refers to those sites that are essential (100% irreplaceable) no matter what the aim. In this instance, there are three such sites (Inner Moray Firth, The Wash and the Ouse Washes). Thus, regardless of the representation target these three sites are always required to satisfy any of the representation targets included in this analysis. Of note and extremely encouraging, is that each of these three wetlands have been classified as SPAs and designated as Ramsar Sites. Nonetheless, the reason for the global irreplaceability of these three sites may simply be that they are amongst the largest of the wetlands in Great Britain (Inner Moray Firth = 2339.2ha, The Wash = 62211.7ha and The Ouse = 2447.3ha; Stroud et al. 2001), and are thus able to support considerably larger numbers of individuals than the other sites (see also Chapter 7).

It is not sufficient, however, simply to classify the sites with the highest irreplaceability as protected areas. Indeed, these sites may not correspond to an optimal solution or actually satisfy the desired representation targets. Rather, the network with the highest overall irreplaceability score (sum of individual site *I* values) or the highest overall mean irreplaceability with the lowest standard deviation should be the top priority network of sites. Should any of these sites become unavailable then if they are replaceable they should be replaced by another site whilst still maintaining an optimal set of sites. Thus, a network approach to irreplaceability as well as for the selection of priority sites is essential rather than looking at individual sites (see Chapters 2-5).

6.5 Conclusions

Algorithms are not a panacea, they are part of the overall planning process not the process itself (Pressey and Cowling 2001). It is possible to integrate various conflicting objectives into the reserve selection framework and multiple data sources can be used. Thus, the planning process itself is made as objective and repeatable as possible. However, the results of systematic site selection algorithms should be used as a starting point for planning and negotiation rather than the final outcome.

CHAPTER 7: MANAGEMENT CONFLICTS

"Acquiring a nature reserve looks like an achievement, but it will come to naught unless the place is looked after properly (Marren 2002)."

As a consequence of an ever-expanding human ecological footprint on natural/semi-natural systems, simply awarding protected status to priority conservation areas is not sufficient to ensure the persistence of biodiversity. Rather, for any conservation planning exercise, it is essential that the appropriate management activities be employed from the outset. However, in most cases, this issue is complicated by the need to account for multiple species on individual sites (Chapter 3.2). Given that species differ in their habitat requirements, it is likely that individual sites will require various contrasting management activities concurrently. Thus, the success of management actions, and ultimately of the network as a whole in supporting viable populations of target species may be determined by the recognition and accommodation of potential conflicts between species in terms of their habitat requirements. This chapter addresses the likely occurrence of management conflicts amongst Annex I and Annex II species occurring on SPA and Ramsar Sites. Following on from this, Chapters 8 and 9 further explore the characteristics of selected migratory waterbird populations, and their implications for the management of protected areas; namely the occurrence of a buffer effect (Chapter 8), and fluctuations in numbers at the local and regional scale (Chapter 9) across wetlands in Great Britain.

Managing wetlands for waterbirds: Are there conflicts between species?

7.1 Introduction

Protecting systematically selected areas of land is a major step towards biodiversity conservation. Although it may not always be the best way to protect species or habitats (Balmford 2002, Pain and Donald 2002), it is certainly one of the most effective and efficient and has, therefore, become one of the most widely adopted measures. Indeed, the designation of priority conservation areas has become a central theme for many national and international conservation strategies. At least 13 international conventions and programmes require the selection of protected areas, including, The World Heritage Programme (161 Natural World Heritage sites in 72 countries; whc.unesco.org/nwhc/pages/home/pages/homepage.htm), The Ramsar Convention on Wetlands of International Importance (1,080 Ramsar Sites in 123 countries; www.ramsar.org) and The Man and Biosphere Convention (393 Biosphere Reserves in 94 countries; www.unesco.org/mab). Furthermore, at least eight additional Conventions and programmes encourage the designation of protected areas, including Article 8 of the Convention on Biodiversity, Article II of the Western Hemisphere Convention and Article X of the African Convention. Despite this excellent progress over the last few decades in recognising nationally and internationally important sites for conservation, a staggering number of protected areas exist in name only (Davey 1998, Hockings and Phillips 1999). Inevitably, without adequate and appropriate management, any conservation gains achieved by the designation of protected areas are likely to be transient (Hockings and Phillips 1999).

The management of protected areas has come to prominence in recent years, particularly as concerns over the poor performance of existing protected areas and the accelerating loss of biodiversity have increased (Hockings and Phillips 1999). This is evidenced by the explicit inclusion of management clauses within international conservation conventions and programmes (e.g. Article 26 of the Convention on Biodiversity). Nonetheless, this sudden proliferation of interest in ensuring the successful management of protected areas can be largely attributed to the growing realisation of the potential benefits to society as a whole rather than simply for conservation purposes. As a result, governments and conservation agencies are, out of necessity, devoting more attention to management issues. In particular, under the Wild Birds Directive (79/409/EEC), all European Union member countries are legally committed to the management of important areas for waterbirds as Special Protection Areas (SPAs). Each country is required not only to identify and classify nationally important wetland areas as SPAs (see Chapter 2), but also to develop and administer management plans at the local site level.

Given the diversity of species, habitats and resources amongst EU countries, no single strategy for the management of SPAs will be universally applicable. Therefore, under the Birds Directive, the precise means by which sites should be managed for the benefit of their waterbird populations has been made the responsibility of each country individually. As a basic guideline, the Directive explicitly states that signatories should take the appropriate measures to ensure that sites maintain a 'favourable conservation status' and that any measures taken do not lead to a deterioration of the current (initial) situation (Article 13). Indeed, failure to show satisfactory evidence of compliance to the various Articles of the Directive may result in a summons to the European Court of Justice (Chapter 3.2). To comply with the various Articles of the Birds Directive, in the UK the Joint Nature Conservation Committee (JNCC) devised and published explicit guidelines pertaining to the management of SPAs in 1999. Included in these guidelines is the stipulation that those sites of national importance for a species (which support more than 1% of the biogeographic population of an Annex II species and 1% of the national population for an Annex I species; see Chapter 2) receive management activities specific to that species. Inevitably, a single wetland site may support nationally important numbers of several species, all of which will be listed as targets for management (Stroud 2002). In addition, those wetlands that regularly support internationally important numbers of individual waterbird species (i.e. 1% of the biogeographic population) are likely to have been designated as wetlands of international importance under the Ramsar Convention on Wetlands of International Importance (Ramsar Sites). Managers of such sites are, therefore, bound by the Convention to maintain these internationally important populations.

Waterbirds as a group are diverse in their wintering habitat requirements. In consequence, where sites are listed as important (nationally or internationally) for multiple species, conflicts of interest in terms of local site management requirements are inevitable (see also Chapter 9). For example, activities aimed at the conservation of one species or assemblage could potentially negatively impact another, such as provisions for diving versus dabbling waterbird species or improving grasslands for waders versus geese (Vickery et al. 1997). Therefore, although site managers are expected to target the populations of all species listed as nationally important at a particular site, conflicting management requirements may mean that it is virtually impossible to provide for all such species on any given wetland.

With this in mind, the aim of this chapter was to identify nationally important inland wetland species most likely directly to conflict in their winter habitat management requirements, and to suggest whether it is theoretically possible to manage individual sites for multiple target species. To address this question, detailed habitat information for a selection of migratory waterbird species common to wetlands in the UK was incorporated into a principal components analysis (PCA). For those species identified as highly contrasting in their habitat requirements, the rates of population size change on wetlands where both species occur together were compared with the rates of change on wetlands where only one of the species occurs to assess the dynamics of potentially conflicting species.

7.2 Methods

Species

Data for 14 species of waterbird common to inland wetlands in the UK were included: little grebe, great crested grebe, mute swan, Bewick's swan, gadwall, mallard, pintail, shoveler, pochard, tufted duck, goldeneye, smew, goosander and coot. These species were those for which sufficient information of consistent quality were available for each environmental variable.

Abundance data

Analyses were conducted using abundance data from WeBS. Data for the period 1985/86-1998/99 were included for those sites classified as SPAs or designated as Ramsar Sites (PS network). For the 14 species separately the maximum number of individuals recorded on each of the WeBS sites between September and March ($N = 1962$) was then determined for each year 1985/86-1998/99.

The PS network is not static and new sites are classified/designated in each year (see Fig. 2.1). These analyses, therefore, excluded all those sites for which the protected status changed over the period 1985/86-1998/99. Thus, the PS network was taken as those sites classified prior to and including 1985/86 ($N = 138$). This means that the numbers of LS included will not necessarily equate to current totals published by the JNCC in the recent SPA review (Table 7.1).

Environmental data

Data were taken from Tucker and Evans (1997), the third volume in BirdLife International's ongoing 'Important birds areas in Europe' programme, encompassing a range of 12 wintering habitat characteristics for each of the 14 waterbird species. Additional habitat data were obtained from Cramp and Simmons (1994), Bruun et al. (1992), Jonsson (1996), and a general internet search for information. Table 7.2 details the means by which each species was scored in relation to the 12 variables individually.

Analyses

A correlation-based principal components analysis (PCA) was performed on the variables listed in Table 7.2 to obtain measures of wintering habitat characteristics that were independent of each other. The components were then interpreted using the component loadings (correlations between the principal component and each original variable).

Table 7.1: The numbers of consolidated sites listed as nationally important for each of the 14 species individually. x No LS.

Species	Number of LS
Little grebe	15
Great crested grebe	19
Mute swan	x
Bewick's swan	18
Shelduck	38
Gadwall	28
Mallard	9
Pintail	31
Shoveler	37
Pochard	12
Tufted duck	7
Goldeneye	14
Smew	x
Goosander	x
Coot	6

Potential conflicts between species were measured in terms of the between-species Euclidean distances in the PC1-PC3 space. The resultant dissimilarity matrix was then subdivided into quartiles, the top 25% of which was taken to be those species most likely to conflict in their management requirements. For those species with the greatest potential for conflict with another species, the rates of population change were compared over the 14 year period 1985/86-1998/99. Comparisons of these rates were made between those PS for which the conflicting species were listed as nationally important (sites supporting ≥1% national population of a species or ≥ 2,000 birds) or not, using the non-parametric Mann-Whitney U test.

For each species individually, the rates of population size change for each site (henceforth termed simply 'rates of change') were calculated for the period 1985/86-1998/99. For each of the 14 species, a set of scatter plots (one for each site) was generated, with the local site population plotted for each year. For each wetland, the rate of change was taken as the regression slope of the individual scatter plots.

Various regression models (Minitab ™ version 13) were tested, however, in each case linear regression resulted in a significantly lower residual sum of squares term and was therefore considered the most suitable. For each wetland, the rates of change were calculated using both the raw (absolute rates) and the $\log_{10}$ local site numbers (relative rates).

Table 7.2: Wintering habitat characteristics included in a principal components analysis. Each species was scored on the 12 axes individually.

Habitat characteristic	Scoring used
Extent of open water	0 = No preference; 1 = Small area; 2 = Small to Medium area; 3 = Medium to Large area; 4 = Large area
Water depth	0 = No preference; 1 < 20m; 2 <20-20/200m; 3 = 20-200m; 4 = 20-200/>200; 5 > 200m
Water flow	0 = No preference; 1 = Standing, 2 = Standing/Slow flow; 3 = Slow flow; 4 = Slow flow/Medium flow; 5 = Medium flow; 5 = Medium flow/Fast flow; 6 = Fast flow
Trophic status	0 = No preference; 1 = Oligotrophic; 2 = Mesotrophic/Oligotrophic; 3 = Mesotrophic; 4 = Mesotrophic/Eutrophic; 5 = Eutrophic
Plant community	1 = Submergent; 2 = Emergent; 3 = Submergent/Emergent; 4 = Submergent/Emergent/Floating; 5 = Emergent/Floating; 6 = Emergent/Floating/Terrestrial; 7 = Emergent/Terrestrial; 8 = Trees
Vegetation height	1 < 10cm; 2 <10/10-100cm; 3 = 10-100cm; 4 = 10-100> 100cm; 5 > 100cm; 6 = Mixed
Plant cover	1 = Nil; 2 < 20%; 3 <20/20-60%; 4 = 20-60%; 5 = 20-60/ >60%; 6 > 60%; 7 = Mixed
Landscape area	0 = No preference; 1 = Small area; 2 = Small to Intermediate area; 3 = Intermediate area; 4 = Intermediate to Large area; 5 = Large area
Landscape features	1 = Fully open landscape; 2 = Mainly open landscape; 3 = Some open landscape; 4 = Either open or mosaic; 5 = Some mosaic habitat; 6 = Mainly mosaic habitat; 7 = Fully mosaic habitat
Food requirements	1 = Vegetation invertebrates; 2 = Aquatic invertebrates; 3 = Seeds; 4 = Aquatic invertebrates/Seeds; 5 = Fish; 6 = Aquatic invertebrates/Fish
Water features	0 = No preference; 1 = clear water; 2 = clear, fresh water; 3 = clear, saline water; 4 = fresh water; 5 = saline water
Dependence on other habitats	0 = No preference; 1 = coastal waters; coastal and inland wetlands; 3 = inland wetlands; 4 = inland wetlands and agricultural lands; 5 = agricultural land; 6 = coastal, inland and agricultural lands

Priority site selection

To determine the implications of including possible conflicts between individual species on a priority site selection procedure, the results of a linear programming algorithm were compared including and excluding these conflicts. For each wetland site included, the total conflict score was taken as the sum of the Euclidean distances for all pairs of species present. For these analyses 1962 sites were included, the same set for each of the 14 species and in all years. These sites are those for which at least 70% of abundance records were available across all years and for each species. Any missing values were imputed using linear interpolation (SPSS version 11), a method similar to that adopted by WeBS for the calculation of national indices (Pollitt et al. 2000, Kershaw and Cranswick 2003).

Chapter 5 concluded that the most effective means by which priority sites for waterbirds should be selected used peak counts and ten years of abundance data within the linear programming algorithm. The approach used here, therefore, incorporated the peak count for each wetland site for each of the 14 species over the ten year period, 1989/90-1998/99.

The site-selection approach used in this chapter is a modified minimum set covering problem (Pressey et al. 1997; Chapter 4). Originally developed for operations research, this mathematical reserve selection method aims to represent all natural features (e.g. species or habitats) a given number of times in the smallest possible area, fewest numbers of sites, or with the lowest overall cost (Pressey et al. 1997; Chapter 4). Typically, analyses of this type have concentrated on the identification of the minimum set of sites required to represent all species at least once. However, for these analyses, complementary networks (Pressey et al. 1993, Cabeza and Moilanen 2001) were obtained using the peaks of abundance for each of the 14 species, with representation targets set for each species individually. Representation targets were derived following the procedure outlined in Chapter 4 (Table 4.1).

Thus, each algorithm used here selects from all possible sites (in this case $N = 1962$) the network that minimises the total conflicts between species and satisfies numerical representation targets calculated for each of the 14 waterbird species individually. Throughout, optimal solutions were obtained using CPLEX™ linear programming software (ILOG 2001).

7.3 Results

Four PCA factors with eigenvalues >1 effectively represented the original 12 environmental variables and together explained >85% of the variation. The first principal component (PC) explained 31.3% of the variation and was composed of plant community type, landscape area and vegetation height. Thus, PC1 describes a gradient from small areas with short, submergent vegetation to large areas with tall vegetation and trees. The second principal component (PC2) explained a further 21.3% of the variation and is positively correlated with water depth and negatively correlated with a dependence on other habitats. PC2, therefore, describes a gradient from shallow waters and a dependence on both coastal and agricultural lands to deeper waters with no dependence on other habitats. The third principal component (PC3) accounts for 14.3% of the variation and can be interpreted as a general measure of landscape features, i.e. representing a gradient from open to mosaic habitat areas. The fourth principal component (PC4) explains a further 11.8% of the variation and can be interpreted as a measure of wetland trophic status. Thus, PC4 describes a gradient from oligotrophic to eutrophic wetlands. The relative positions of each of the 14 species with respect to PC1/PC2 and PC1/PC3 space are shown in Fig. 7.1.

The positions of each of the 14 species seem to make intuitive sense in both the PC1/PC2 and the PC1/PC3 phase space. In particular, species often observed on the same wetlands (e.g. goosander and smew, coot and tufted duck, gadwall and pintail) cluster together.

From the PCA, 23 species pairs were identified as highly likely to be conflicting in terms of their habitat requirements (Table 7.3). These species pairs were those where the Euclidean distance between PC1, 2 and 3 was >2.88 (75% quartile), and are henceforth referred to as 'conflicting species pairs'. The largest difference between a pair of species was for pintail/Bewick's swan (Euclidean distance of 4.14), followed by goosander/mallard (Euclidean distance of 3.76). Conversely, for 22 species pairs the distance was <1.64 (Table 7.3). These species are those considered least likely to conflict in their management requirements (25% quartile). The minimum Euclidean distances between species were for coot versus tufted duck (0.41) and also for tufted duck versus pochard (0.54).

For each of the 23 conflicting species pairs, the numbers of sites where both species were listed as nationally important (LS) ranged from two to 15 (Table 7.4). Specifically, the greatest number of LS shared was for pintail and great crested grebe (15 sites). For 13 species pairs either one or both the species have no LS (i.e. pairs containing mute swan, smew or goosander). Correlations between the rates of population size change (relative and absolute) were significant for five of the 23 conflicting species pairs (mute swan/gadwall, gadwall/goldeneye, mallard/gadwall, mallard/goosander, goldeneye/goosander; Table 7.4). Further, each of these statistically significant correlations was positive. Thus, for these species pairs where one species has a positive rate of increase so does the other.

Considering the relative rates of change for the most likely conflicting species pairs only six of the 23 pairs showed significant differences between sites where a potentially conflicting species was listed as nationally important and other protected sites. Specifically, for great crested grebe/gadwall, mute swan/gadwall, pintail/Bewick's swan, smew/pintail, goosander/mallard and goosander/goldeneye the relative rates of population size change for sites where the conflicting species was an LS were significantly faster (i.e. gaining individuals at a faster rate) than for the non-LS wetlands (Table 7.5). In each case, the relative rates of population size change for the first of these species pairs (great crested grebe, mute swan, pintail, smew, goosander) were grouped by the second species (Bewick's swan, gadwall, mallard, pintail, goldeneye).

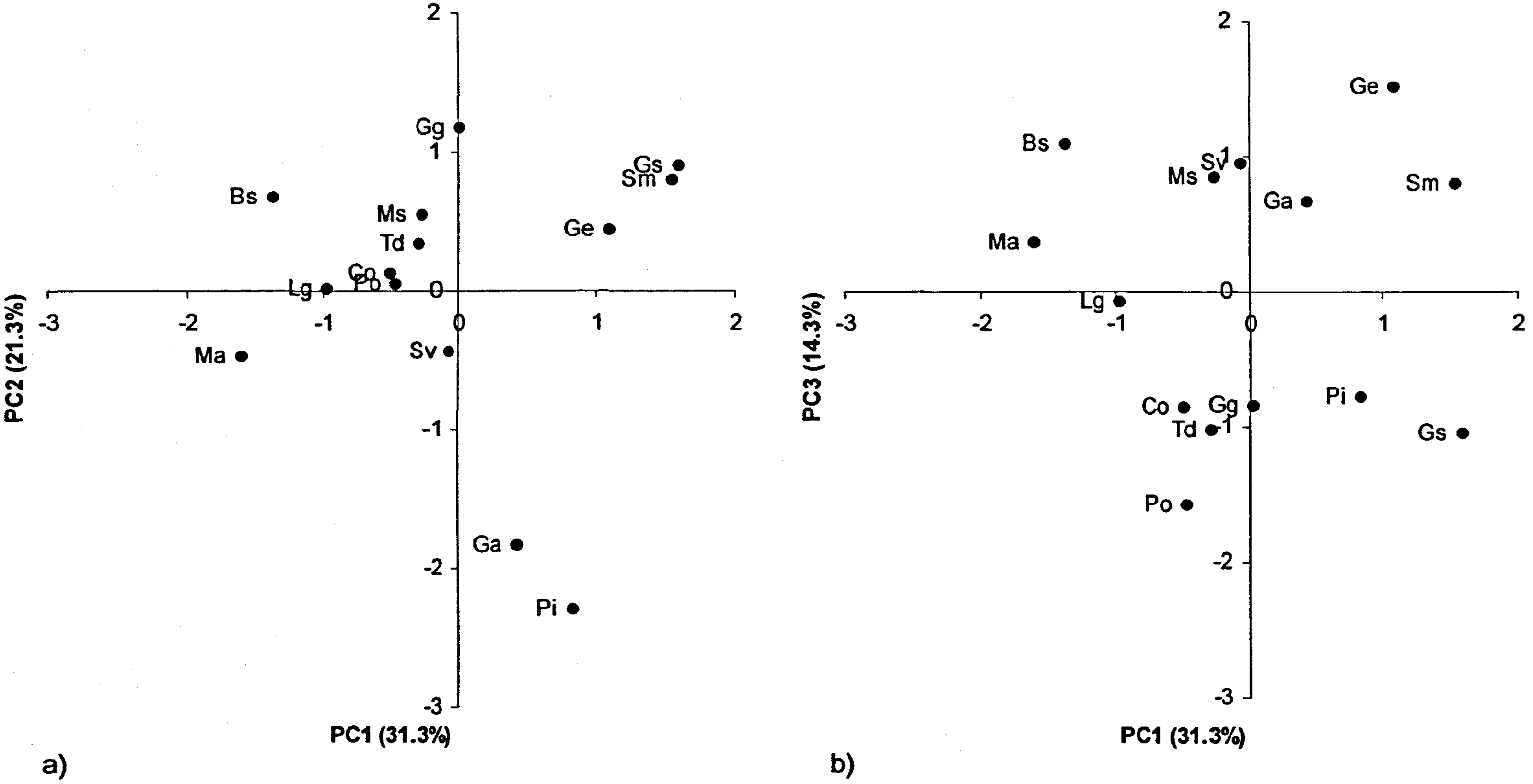

Figure 7.1: The distribution of each of the 14 species with respect to a) principal component in relation to (PC) 1/PC2 and b) PC1/PC3 phase space. Lg = little grebe, Gg = great crested grebe, Ms = mute swan, Bs = Bewick's swan, Ga = gadwall, Ma = mallard, Pi = pintail, Sv = shoveler, Po = pochard, Td = tufted duck, Ge = goldeneye, Sm = smew, Gs = goosander, Co = coot.

In particular, at 75.0% of sites where gadwall was listed as nationally important (LS), the numbers of great crested grebe increased over the period 1985/86-1998/99, compared with 50.9% of sites where gadwall was not listed (Fig. 7.2a).

Table 7.3: Euclidean distance between principal components 1 to 3 for each pair of species.

	Little grebe	Great crested grebe	Mute swan	Bewick's swan	Gadwall	Mallard	Pintail	Shoveler	Pochard	Tufted duck	Goldeneye	Smew	Goosander
Little grebe													
Great crested grebe	1.71												
Mute swan	1.28	1.82											
Bewick's swan	1.40	2.43	1.14										
Gadwall	2.44	3.40	2.49	3.12									
Mallard	0.91	2.61	1.76	1.38	2.47								
Pintail	3.02	3.57	3.45	4.14	1.57	3.25							
Shoveler	1.44	2.41	1.01	1.71	1.51	1.64	2.70						
Pochard	1.58	1.42	2.47	2.88	3.06	2.30	2.80	2.60					
Tufted duck	1.22	0.92	1.88	2.40	2.85	2.07	2.87	2.13	0.63				
Goldeneye	2.64	2.69	1.51	2.50	2.51	3.07	3.58	1.56	3.47	2.89			
Smew	2.78	2.27	1.82	2.92	2.86	3.42	3.55	2.03	3.19	2.63	0.92		
Goosander	2.90	1.62	2.68	3.67	3.44	3.76	3.31	2.93	2.29	1.98	2.66	1.86	
Coot	0.93	1.17	1.77	2.21	2.65	1.75	2.76	1.94	0.71	0.33	2.87	2.71	2.25

On average, where gadwall was listed as nationally important (LS) the absolute rates of change for great crested grebe were significantly greater with a mean (±SE) rate of 1.29 (±0.71) birds per year compared with 0.64 (±0.35) birds per year for sites where gadwall was not an LS. Further, 75.5% of the total individuals lost per year ($N = 65.43$ birds per year) were from these latter sites (Fig. 7.2b). A similar pattern was shown for the mute swan/gadwall pair, the pintail/Bewick's swan pair and the goosander/mallard pair (Fig. 7.2a). Indeed, for mute swan, although there was little difference in the percentage of sites with positive rates of change between sites where gadwall was listed and sites where it was not listed, 82.2% of the absolute numbers of individuals lost each year ($N = 77.1$ birds per year) were from the latter sites (Fig. 7.2b).

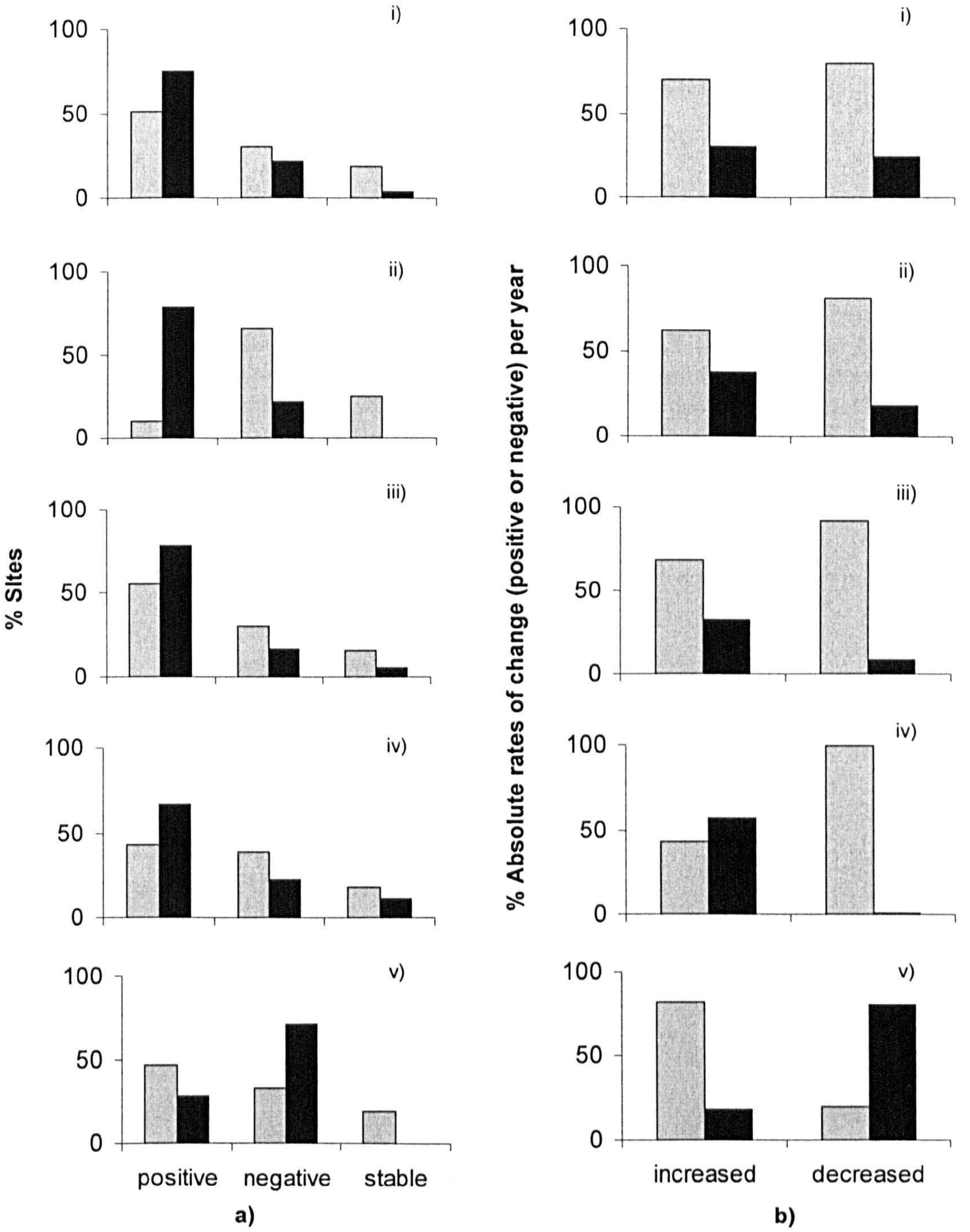

Figure 7.2: Bar charts representing a) the % sites for a species where numbers have increased (positive rates), decreased (negative rates) or remained stable (no change) where the conflicting species is either listed as nationally important (black bars) or sites where the conflicting species is not listed as nationally important (grey bars) and b) the % of the absolute rates of population size change from sites where the conflicting species has increased or decreased where the conflicting species is either listed as nationally important (black bars) or not listed as nationally important (grey bars) for i) great crested grebe/gadwall, ii) mute swan/gadwall, iii) pintail/Bewick's swan, iv) goosander/mallard and v) goosander/goldeneye.

Further, on average, the relative rates of change for mute swan were greater on those sites listed as nationally important for gadwall LS 4.83 (±1.64) birds per year, compared with 1.64 (±0.64) birds per year on other PS wetlands not listed for gadwall. Similarly, 92.1% of the annual losses for pintail were from those PS where Bewick's swan was not listed as nationally important (Fig. 7.2).

Table 7.4: The 23 species pairs most likely to be in conflict (taken as those pairs in with a Euclidean distance in the top 25% quartile), the number of LS shared by the pair, and the Spearman's rank correlation coefficient (r_s) between the absolute rates of population size change amongst all protected sites (SPAs and Ramsar Sites) for the two species. Relative rates are not reported as the Spearman's rank correlations were identical to those using absolute rates. *P <0.05, **P <0.01. x Either one or both species of the pair have no official LS.

Species a	Species b	LS shared	r_s
Little grebe	Pintail	13	0.016
Little grebe	Goosander	x	0.148
Great crested grebe	Gadwall	9	0.123
Great crested grebe	Pintail	15	0.129
Mute swan	Pintail	x	-0.121
Bewick's swan	Gadwall	9	0.043
Bewick's swan	Pintail	6	-0.107
Bewick's swan	Pochard	8	0.058
Bewick's swan	Smew	x	-0.104
Bewick's swan	Goosander	x	-0.123
Gadwall	Pochard	8	0.343**
Gadwall	Goosander	x	0.148
Mallard	Pintail	6	0.157
Mallard	Goldeneye	4	0.240**
Mallard	Smew	x	0.067
Mallard	Goosander	x	0.235**
Pintail	Goldeneye	6	0.046
Pintail	Smew	x	0.171*
Pintail	Goosander	x	0.017
Shoveler	Goosander	x	0.089
Pochard	Goldeneye	4	0.332*
Pochard	Smew	x	0.138
Tufted duck	Goldeneye	2	0.246**

Table 7.5: Mann-Whitney U comparisons for each of the 23 species pairs most likely to be in direct conflict in terms of their habitat requirements. Comparisons were made between the relative rates of population size change for sites where the species potentially in conflict is listed as nationally important (LS) and other non-LS wetlands. In all cases, comparisons of the relative rates of change for each species (rows) were made between those sites where the conflicting species (columns) was listed as nationally important or not. The same comparisons using the absolute rates of change were not made given the confounding effect of population size. *$P < 0.05$, **$P < 0.01$, ***$P < 0.001$.

	Little grebe	Great crested grebe	Bewick's swan	Gadwall	Mallard	Pintail	Shoveler	Pochard	Tufted duck	Goldeneye
Little grebe						1584.0				
Great crested grebe				1139.0*		1586.0				
Mute swan				1082.0*		1430.0				
Bewick's swan				1274.0		1531.5		698.5		
Gadwall		1012.0	868.0					707.0		
Mallard						1434.0				641.5
Pintail	756.5	906.5	624.0**		489.0	.				707.5
Shoveler										
Pochard			978.0	1453.5						758.0
Tufted duck										663.5
Goldeneye				1338.0		1583.5		537.5	388.5	
Smew			935.0	1427.0	545.5	1200.0*				
Goosander	876.0			1491.0	336.0*	1614.0	1762.0			618.0*

Table 7.6: Mann-Whitney U comparisons for all non-conflicting species pairs (i.e. excluding the 23 species pairs identified as most likely to be in conflict shown in Table 7.5). In all cases, comparisons of the relative rates of change for each species (rows) were made between those sites where the conflicting species (columns) was listed as nationally important or not. The same comparisons using the absolute rates of change were not made given the confounding effect of population size. *$P < 0.05$, **$P < 0.01$, ***$P < 0.001$.

	Little grebe	Great crested grebe	Bewick's swan	Gadwall	Mallard	Pintail	Shoveler	Pochard	Tufted duck	Goldeneye	Coot
Little grebe		1063.0	892.0	2190.5	561.5		1735.5	572.5	327.0	722.5	240.0
Great crested grebe	607.0*		793.0		563.5		1241.5**	540.5	218.0*	831.5	206.0
Mute swan	867.5	1032.0	788.5		268.0**		1346.5*	397.5**	319.5	769.0	313.5
Bewick's swan	593.0*	1043.0			476.5		1318.0**		428.0	748.0	386.0
Gadwall	769.0				510.0	1302.0	1314.0*		361.0	842.0	
Mallard	708.0	1032.0	737.5*	1176.5			1495.5	708.0	399.0		328.0
Pintail				1511.5			1589.5	704.5	424.5		373.5
Shoveler	854.0	905.5	689.5*	1278.5	536.0	1456.5		576.5	382.5	599.5	298.5
Pochard	878.5	1023.5			507.0	1347.5	1543.5		377.0		293.0
Tufted duck	844.0	1050.5	897.5	1319.5	460.0	1532.5	1531.0	670.0			244.0
Goldeneye	867.0	1011.5	1046.0		455.0		1556.0				352.5
Smew	526.5**	723.0*				1200.0*	1697.5	623.5	350.0	636.0	230.0
Goosander		1041.0	870.0					632.5	405.5		293.5
Coot	882.0	875.0	716.5*	1174.0	287.5*	1230.5*	1543.0	412.0*	299.0	835.5	

Conversely, for the smew/pintail and the goosander/goldeneye conflicting species pairs, the relative rates of population size change for smew and goosander were significantly slower on wetlands where the conflicting species (pintail or goldeneye) was listed as nationally important compared with other PS wetlands. Specifically, relative rates of change for goosander were significantly lower for the goldeneye LS wetlands compared with other, non-goldeneye LS (-9.23 (±8.94) birds per year compared with 0.007 (±0.16) birds per year). Thus, the numbers of individuals increased at a faster rate on those sites not listed as nationally important for goldeneye. Further, goosander relative rates of change were negative (i.e. losing individuals) for 71.40% of the goldeneye LS, which accounted for 80.6% of the total individuals lost in each year (N = 170.75 birds per year; Fig. 7.2). Similarly, the relative rates of change for smew were significantly lower on PS where pintail was listed as nationally important (-0.004 (±0.16) birds per year) compared to sites where pintail was not listed (0.002 (±0.02) birds per year). Furthermore, smew rates of change were negative for 58.06% of the sites where pintail is listed as nationally important (LS). In addition, 92.30% of the individuals gained in each year (N = 6.85 birds per year) were from those sites where pintail was not listed as nationally important.

Considering all other species pairs (i.e. other than the 23 conflicting species pairs), five species displayed a negative relationship between the population dynamics of one species on another (great crested grebe/little grebe, Bewick's swan/little grebe, Bewick's swan/shoveler, smew/great crested grebe and smew/little grebe; Table 7.6). Thus, the relative rates of change for the first species (great crested grebe, Bewick's swan and smew) were significantly slower on sites where the second species (little grebe, shoveler and great crested grebe) was listed as nationally important compared with other PS wetlands. In addition, for 12 pairs of species, the reverse was true. For great crested grebe/shoveler, great crested grebe/tufted duck, mute swan/mallard, mute swan/pochard, Bewick's swan/shoveler, gadwall/shoveler, mallard/Bewick's swan, shoveler/Bewick's swan, coot/Bewick's swan, coot/mallard, coot/pintail and coot/pochard the rates of change for the first species were significantly faster (gaining individuals at a faster rate) where the second species was listed as nationally important.

Priority site selection

Only one minimum set solution, comprising 78 sites, was feasible when conflict scores for each of the 14 species were included within the priority site-selection algorithm (Fig. 7.3). By contrast, 60 minimum sets, each comprising 59 sites, were generated without the inclusion of these conflicts. Amongst these 60 solutions, 39 sites were 100% irreplaceable (i.e. they occurred in 100% of the optimal solutions) and 99 sites were included in at least one optimal solution (Fig. 7.4). Of these 99 sites, 50 were shared by the conflicts solution (Fig. 7.5). However, six sites were unique to the conflicts solution (Fig. 7.5).

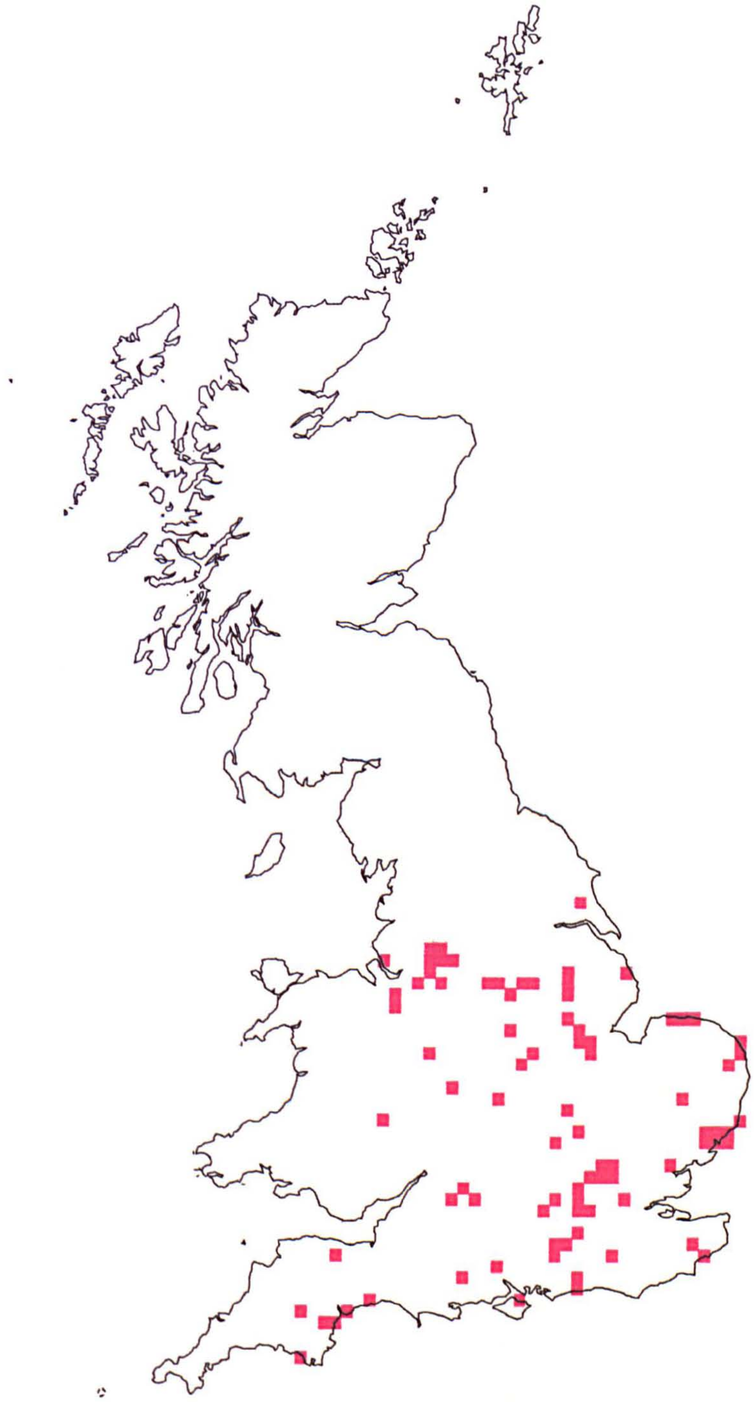

Figure 7.3: The distribution of sites (red squares) selected by the single solution for the linear programming algorithm incorporating conflict scores for each wetland site.

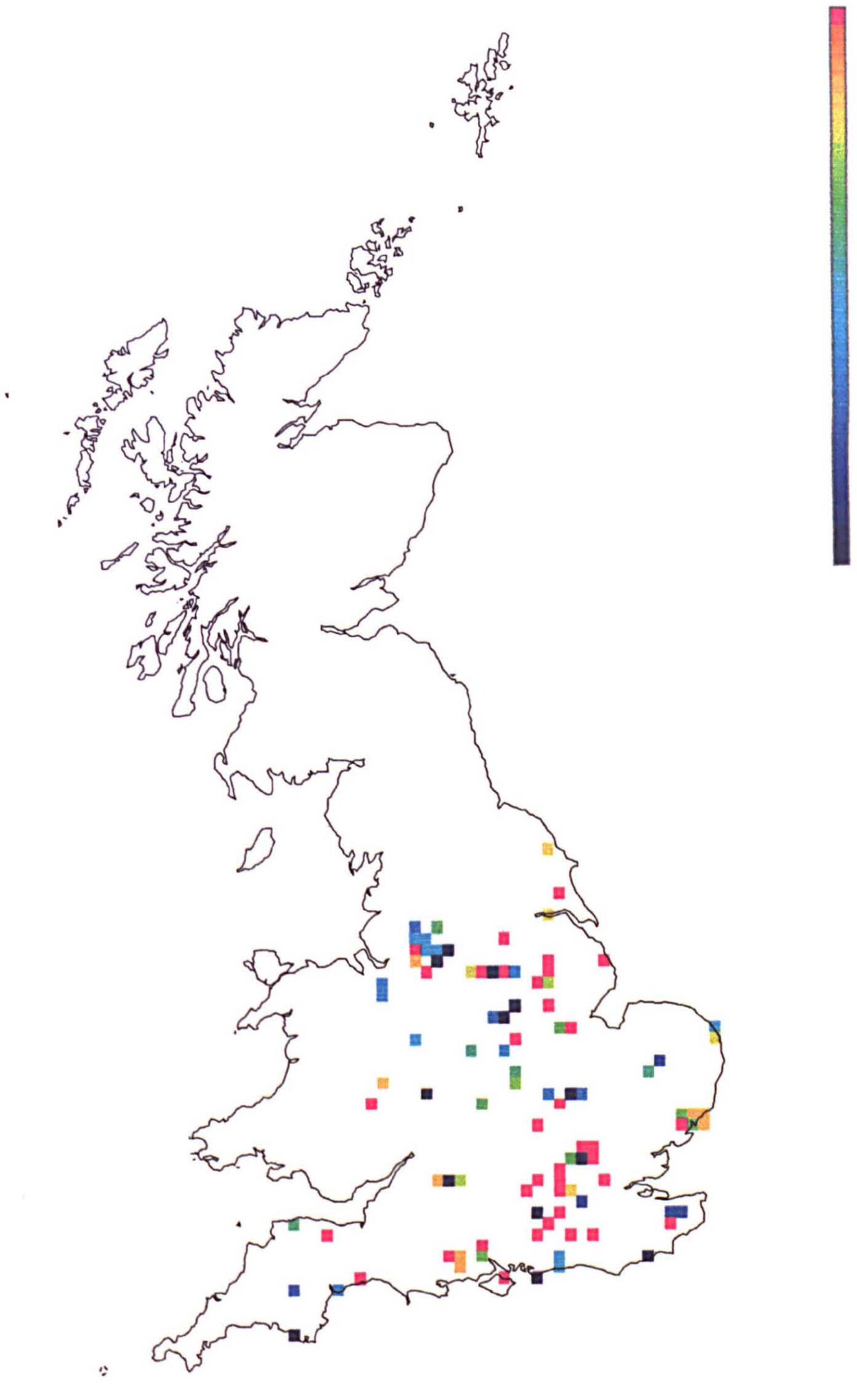

Figure 7.4: The distribution of all sites included in at least one of the optimal solutions using the linear programming algorithm without conflicts included. The equal interval scale spans from the red squares indicating 100% irreplaceability, to the navy blue squares indicating those sites included in only one of the 60 optimal solutions.

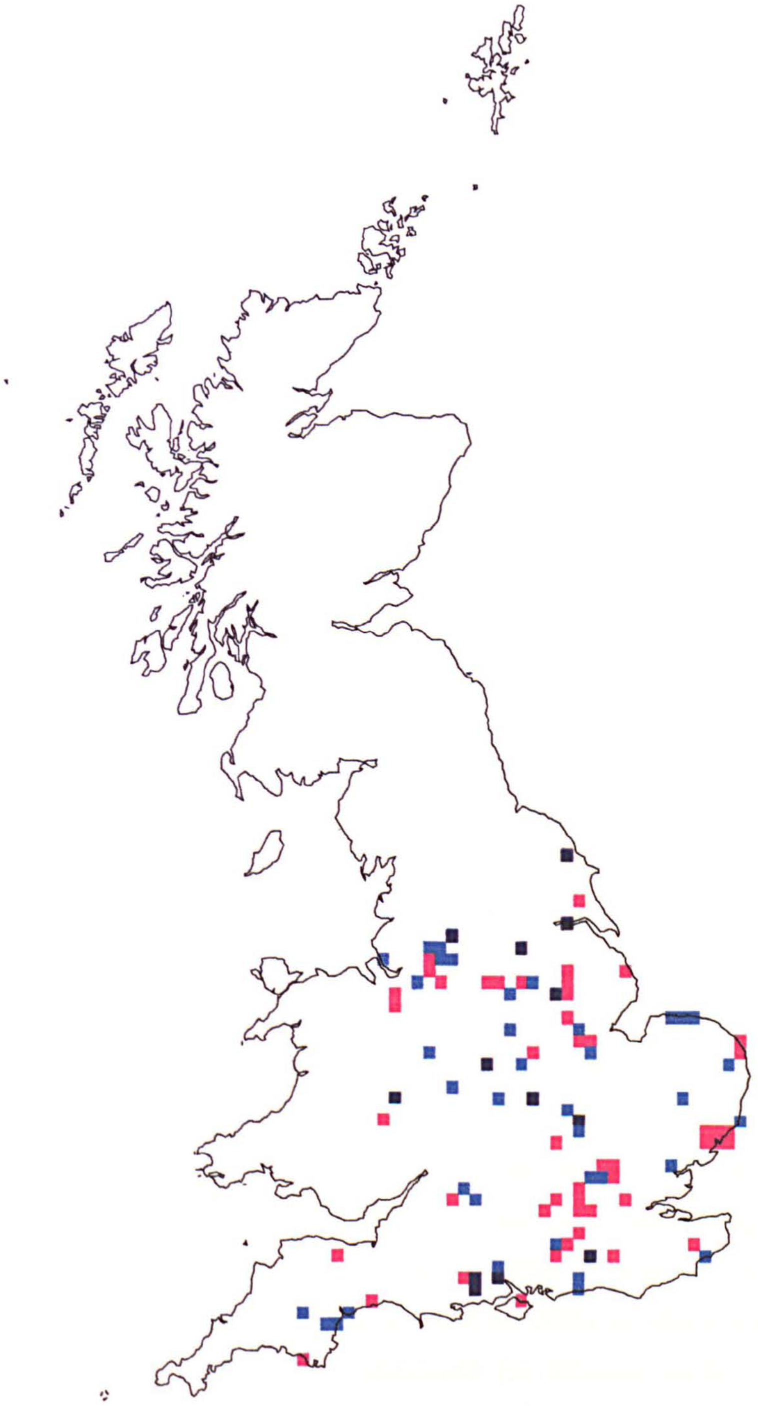

Figure 7.5: The distribution of all sites included by the simple minimum set algorithm (navy blue squares), the algorithm incorporating species conflicts (red squares) and those sites shared by both algorithms (pale blue squares).

7.4 Discussion

Given the limited resources available for conservation purposes, conflicts of interest over individual protected areas are perhaps inevitable. Such conflicts will likely originate both as a result of the protected area designation process and also through the design and application of individual species management plans. First, many protected areas in the UK have been awarded a multitude of conservation designations. For example, The Wash is classified as a National Nature Reserve (NNR), a Ramsar Site, a Special Protection Area (SPA), a Site of Special Scientific Interest (SSSI), a Special Area of Conservation (SAC) and forms part of the North Norfolk Area of Outstanding Natural Beauty (AONB) and the North Norfolk Biosphere Reserve. Each of these designations individually requires that the appropriate measures be taken to maintain the integrity of the site with respect to their particular conservation objectives. For example, as a Ramsar Site, The Wash is necessarily managed for internationally important wintering waterbird numbers, by contrast, as an AONB or NNR the area will be managed to maintain overall site integrity and diversity, incorporating tourism and the requirements of various stakeholders, including local human populations.

Second, aside from the complications arising from multiple conservation designations, for those sites classified as SPAs/Ramsar Sites for several waterbird species, given the diverse range of habitat requirements of waterbirds common to wetlands in the UK, conflicting management requirements are to be expected. For example, The Wash qualifies as an SPA under Article 4.1 of the Birds Directive by supporting populations of European importance during the breeding season (common tern (*Sterna hirundo*), little tern (*Sterna albifronsi*) and marsh harrier (*Circus aeruginosus*)) and during the winter season (avocet (*Recurvirostra avosetta*), bar-tailed godwit (*Limosa lapponica*), golden plover (*Pluvialis apricaria*) and whooper swan). The site also qualifies under Article 4.2 of the Directive by supporting two passage and 11 migratory species, including black tailed godwit (*Limosa limosa islandica*), curlew (*Numenius arquata*), dark-bellied brent goose and pintail. Finally, The Wash qualifies under Article 4.2 by regularly supporting more than 20,000 waterbirds (Stroud et al. 2001).

Previous analyses have identified conflicts between the habitat requirements of wintering waterbird species. First, for waders and geese species on grazing marshes in Norfolk, Vickery et al. (1997) identified a likely conflict in their individual habitat requirements. However, they also concluded that it could be easily resolved through the improvement of grasslands for geese during the winter and for breeding waders in the summer. Second, Sutherland and Allport (1994) identified possible conflicts between overwintering bean geese and wigeon on estuaries in the Yare valley, Norfolk. They conclude that to maintain the current bean goose population would require a reduction in the numbers of wintering wigeon in the area. Nonetheless, the results of the present analysis provide little evidence for conflicts between species on protected areas for waterbirds (SPAs and Ramsar Sites). Indeed, although 23 species pairs were identified as most likely to be in conflict in terms of their habitat management requirements, only five showed significantly different rates of change for one species dependent on whether the other was listed as nationally important (LS). This lack of evidence of conflicts between Birds Directive target species is particularly encouraging, especially given the myriad of possible constraints on the management of protected areas. Further, for four of the five species pairs, the rates of change were significantly faster (i.e. species increasing in numbers across sites at a faster rate) for a species where the conflicting species was listed as nationally important.

One possible reason, however, for the positive impact of listing gadwall on great crested grebe, mute swan and Bewick's swan on pintail, and mallard on goosander as nationally important is simply that these protected sites are not actually being managed for individual species. Indeed, it is likely that the focus of local site management is the overall waterbird assemblage rather than individual species. In this respect, for gadwall 16 of the 19 LS were classified for their overall waterbird assemblage in addition to their nationally important gadwall aggregations. Similarly, 16 of the 18 LS for Bewick's swan and all 13 LS for mallard were also classified for their overall waterbird assemblage (Stroud et al. 2001).

By contrast, only the goosander/goldeneye conflicting species pair demonstrated a potential negative impact of one species on the other. Perhaps the most influential factor is that goosander feed to a large extent on fish, whereas aquatic invertebrates provide the main source of food for goldeneye (Tucker and Evans 1997). Fish are an important food source for certain waterbird species, including goosander, however they are also a major competitor with waterbirds (Giles 1992). In particular, fish eat aquatic invertebrates, decrease aquatic plant growth and increase turbidity. They are also thought to be instrumental in the initiation and continuation of algal contaminated communities (Andrews 1995). In light of the potentially deleterious impacts of fish on aquatic plants and invertebrates, at sites where the target species do not require a supply of fish, numbers are more likely to be tightly controlled in order to provide more food for the target species.

Considering the remaining 18 conflicting species pairs, there may, however, be several reasons for the lack of evidence of conflicts on wetland protected areas. First, SPAs may simply be the largest sites. In consequence, because larger areas are more likely to incorporate a greater habitat and resource diversity than smaller areas (Gaston et al. 2002), such sites can sustain a variety of management activities directed towards individual species. Thus, sections of a particular lake or inland waterbody can be allocated different management regimes specific to various different target species individually.

Second, because waterbirds are only likely to occur on those sites where the habitat is suitable, species may in fact only be coexisting on the largest sites with a sufficient diversity of habitats. The populations of each species will essentially be regulated by the amount of available habitat. However, because the numbers of individuals on many sites are on the increase (see Chapter 3.2), this would suggest that the species are not, as yet, in direct conflict.

A third possibility is that sites are not in fact being managed for individual species. Therefore, assuming a site was suitable for a species in the first instance, there will be no new conflicts arising from a modification of the habitat (i.e. through deepening of lakes, planting of emergent vegetation, or vegetation clearance).

Fourth, species may tolerate a wider range of factors than allowed for by the PCA. For example, species can occur on sites with characteristics outside their general range (i.e. a wider range of vegetation heights and a range of vegetation compositions), although, this is unlikely to extend to characteristics such as water depth and trophic status.

Fifth, birds do not confine themselves to a single site in any one winter season (Moser 1987, Goss-Custard and Durrel 1990, Skagen and Knopf 1993, Rehfisch et al. 1996, Farmer and Parent 1997, Pradel et al. 1997, Madsen et al. 1998). Individuals use sites for a variety of purposes (i.e. for feeding, roosting or loafing). In consequence, the habitat characteristics required for roosting areas are unlikely to be the same as for feeding areas. For example, although dabbling ducks such as mallard and shoveler generally feed in water up to 0.4m in depth, whereas diving species such as little grebe, goldeneye and goosander feed in water from 0.5 to 2.5m (although some mollusc and fish feeders go deeper), deep water is preferred by both groups for roosting (Andrews 1995). Thus, the time of day/month/season at which WeBS counts were carried out will inevitably affect which site individual birds are likely to be on. In consequence, if a site is being actively managed for one species, another species may not be adversely affected if it is not using it for feeding/roosting.

Non-conflicting species pairs

Seven species (great crested grebe, mute swan, Bewick's swan, gadwall, mallard, shoveler and coot) were potentially positively affected by the listing of another species as nationally important. For these species, given their similar habitat requirements, it is likely that management actions addressed at one will ultimately benefit the other. Nonetheless, this is not true if the order of the pairs is reversed, for example, the numbers of great crested grebe increased at a faster rate where shoveler was listed as nationally important, however there was no significant difference in the rates of change for shoveler where great crested grebe was listed. This would, therefore, suggest either that the management employed for shoveler is less specific than that employed for sites listed as nationally important for great crested grebe (i.e. management is aimed directly at the overall waterbird assemblage rather than specifically for shoveler), or that great crested grebes can tolerate a wider range of habitat features than shoveler.

Alternatively, rather than resources being the only limiting factor, it may be that certain species show more inter-specific aggression than others, which may force another species onto peripheral areas or alternative feeding areas. This phenomenon relates to the despotic distribution concept introduced by Fretwell (1972), where individuals restrict the access of others to resources through aggression. This is in direct contrast to the ideal free distribution which states that the spatial distribution of individuals after settling relates exactly to the distribution of resources. The available evidence is, however, restricted to territory defence during the breeding rather than the wintering season.

A further explanation relates to differential tolerance to disturbance for individual waterbird species. Indeed, one species may be less able to tolerate disturbance, therefore the possible negative impact shown may simply reflect a tolerance hierarchy for these species. For instance, work by Tuite et al. (1984) demonstrated a gradient of susceptibility to disturbance from recreation/tourism for a selection of waterbird species on inland waters in England and Wales. They concluded that mallard were the least susceptible, whereas goldeneye were the most susceptible to disturbance. Similarly, Klein et al. (1995) demonstrated that some areas used for recreational purposes were devoid of certain species on lakes in the Ding Darling National Wildlife Refuge, Florida. This explanation, however, assumes that the LS for one species (i.e. the conflicting or grouping species) are by chance used for recreation/tourism more than those for the other species.

Priority site selection

The inclusion of conflicts into the priority site selection algorithm not only increased the numbers of sites included in the minimum set, but also decreased the flexibility of the final network of priority sites (i.e. only one solution as opposed to 60 without conflicts included). This is indicative of an overall cost to reserve network efficiency through the inclusion of potential species conflicts in the selection process. Nonetheless, this analysis provides an example of the possible implications of the inclusion of possible conflicts on area selection. Although only one optimal solution was feasible when species conflicts were incorporated into the selection algorithm, this approach could be particularly valuable for real-world conservation planning. Indeed, the presentation of a single solution incorporating such conflicts does not mean that this set must be rigidly followed; rather it should be used to guide negotiations (see Chapter 6), and highlights the need to consider potential conflicts carefully. Additionally, allowing for species conflicts at the management level will enable provisions to be made to ensure potential conflicts do not relate to actual conflicts to the detriment of one or more target species.

7.5 Conclusions

Although various potential conflicts between the 14 species analysed were identified, there is little evidence of any detrimental effects amongst the PS network wetlands. Nonetheless, recognition of these potential conflicts is essential throughout the management process to ensure the populations of all EU target species remain suitably and indeed adequately protected into the future. In terms of the management of SPAs and Ramsar Sites these results are encouraging, given that, for the majority of species there is no evidence for a significant negative impact of the listing of one species as nationally important on another. Of concern are those sites listed as nationally important for goldeneye where goosander rates of change are significantly lower than for other sites, particularly as the level of protection afforded to goosander is less than for other species in that this species was omitted from the original SPA selection process (Chapters 2 and 3).

CHAPTER 8:

The buffer effect and migratory waterbirds on British wetlands

8.1 Introduction

A common and widely used concept in ecology is that of carrying capacity, which can be defined as the maximum number of individuals a given ecosystem/habitat/site can sustain indefinitely without degrading the environment or imposing intolerable density-dependent restrictions on individuals or the population as a whole (Brown 1969, Fretwell and Lucas 1970). Theory predicts that should the sustainable limits of a particular area be exceeded, either in relation to resource depletion or the availability of un-occupied territories, further increases in population size will be constrained by the resource in shortest supply. Costs may then be incurred either at the level of the whole population, through a reduction in the average fitness of all individuals due to decreased resource availability, or disproportionately at the level of the individual, through antagonistic encounters with dominant conspecifics (Halama and Duesser 1994, Ferrer and Donazar 1996).

An extension of carrying capacity is the buffer effect. Conditional on there being demographic costs of inhabiting poor quality sites, this predicts that where reproductive success and survivorship vary between potential habitats, sites will be sequentially filled according to a preference hierarchy (Fretwell and Lucas 1970, Moser 1988, Gill et al. 2001). The expectation is that where population densities are low, poor quality sites will be avoided, individuals instead favouring sites of higher quality. In this instance, the variance in fitness between individuals is expected to be low (Ferrer and Donazar 1996). Sustained population increase will, however, force a growing proportion of individuals onto these poorer quality sites, thereby increasing the variance in fitness within the population (Brown 1969, Ferrer and Donazar 1996). Thus, the buffer effect predicts that once favoured sites reach saturation, numbers on the less-suitable/poorer quality sites will show a greater rate of increase compared with those on favoured sites (Gill et al. 2001) and is represented graphically as a significant negative correlation between initial population sizes and rates of change at each site (Fig. 8.1). In essence, following sustained population increases, population sizes on preferred sites will be buffered over time by those on the less-suitable habitats (Gill et al. 2001, Vickery 2001).

Evidence in support of the buffer effect is readily available in the literature for many species, although this is generally restricted to analyses at small spatial scales. Examples include aphids (Whitham 1978), side-blotched lizards (Calsbeek and Sinervo 2002), Spanish imperial eagles (Ferrer and Donazar 1996), oystercatchers (Goss-Custard 1980), dunlin (van de Have et al. 1984), great tits (Krebs 1970) and mice (Morris 1989, Halama and Duesser 1994). However, two studies also provide evidence at a national-scale, for grey plover (Moser 1988) and black-tailed godwits (Gill et al. 2001) wintering on estuaries in Great Britain.

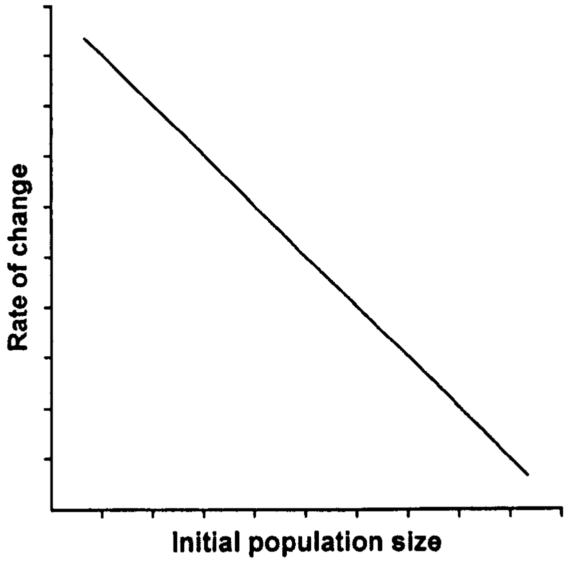

Figure 8.1: Theoretical model of a buffer effect showing a negative relationship between the initial population size (five-year mean) and the rate of change for each individual site.

In Great Britain, the existence of a buffer effect would have significant implications for the selection and management of internationally and nationally important wetlands as Ramsar Sites and Special Protection Areas (SPAs) for waterbirds and, in particular, for the use of 1% thresholds for site-selection. Briefly, under this criterion, a site is considered to be of conservation importance if it regularly holds greater than 1% of either the national or the biogeographic population of a target species (JNCC 1999, Ramsar Convention Bureau 2002). Inevitably, however, should national numbers continue to increase (as they have been doing for many wetland species; Kershaw and Cranswick 2003), these 1% threshold levels will represent progressively larger numbers of individuals. Therefore, once populations on preferred sites reach capacity, they will decrease in importance relative to these threshold levels, and may no longer satisfy Ramsar or SPA qualification criteria. Of concern, is that the spill-over sites, whilst being of numerical importance, are likely to represent sink populations and may be the first to lose individuals should national populations subsequently decline (Moser 1988).

In this chapter, I analyse population sizes for 19 species of migratory waterbird on wetland sites in Great Britain to test the generality of any evidence of buffer effects at this spatial scale. For all these species there has been a sustained temporal increase in the national population (see Pollitt et al. 2000), providing a circumstance in which the effects are likely to be displayed.

8.2 Methods

These analyses were conducted using WeBS data for the years 1980/81 to 1998/99 inclusive, for 19 species of regularly occurring migratory waterbird considered to be well represented by WeBS, namely: mute swan, Bewick's swan, whooper swan, Canada goose, dark-bellied brent goose, shelduck, wigeon, gadwall, teal, pintail, shoveler, tufted duck, goldeneye, red-breasted merganser and goosander. In addition, because counts began at a later date for some species data for 1982/83-1988/99 were included for great crested grebe and coot, 1985/86-1998/99 for little grebe and 1986/87-1998/99 for cormorant.

Table 8.1: Numbers of sites included in these analyses for each of the 19 species

Species	Sites
Little grebe	412
Great crested grebe	308
Cormorant	421
Whooper swan	96
Bewick's swan	196
Mute swan	138
Canada goose	129
Dark-bellied brent goose	61
Shelduck	118
Gadwall	125
Pintail	130
Shoveler	135
Wigeon	96
Teal	140
Tufted duck	143
Goldeneye	137
Goosander	136
Red-breasted merganser	107
Coot	343

Site counts

For each year, the total bird count at each wetland site was calculated for each species using two methods, peak annual counts (the accepted method for WeBS) and bird-days (following Gill et al. 2001). For the former method, the annual total per site is taken as the maximum count at a site from monthly winter counts (September to March inclusive). Alternatively, the numbers of bird-days are equal to each monthly count multiplied by the number of days in that month. These monthly totals were then summed to give an overall bird-day count for each individual site. Given that the results from the two methods were essentially identical, only those using bird-days have been reported here. For each species only those sites with at least three monthly counts for all years were included. Thus, the percentage of missing values was kept to a minimum. The remaining missing values were imputed by linear interpolation (SPSS, version11), the method adopted by WeBS (Pollitt et al. 2000; Kershaw & Cranswick 2003). In addition, those sites visited in all years but where no individuals of a species were recorded were also excluded from the analysis.

To overcome the problem that change in the protected status of a site (non-SPA to SPA) may, because of management activities, affect local population sizes, any site changing status was excluded. Thus, for the SPA sites only those classified as SPAs prior to and including 1980/81 (or 1982/83-1998/99 for great crested grebe and coot, 1985/86-1998/99 for little grebe and 1990/91-1998/99 for cormorant) were included. The numbers of wetland sites included for each species are shown in Table 8.1. Given that WeBS counts in Northern Ireland did not begin until 1985 these analyses use data from Great Britain only.

National Numbers

As all WeBS sites have not necessarily been counted in all years it was not possible simply to sum the individual site totals to obtain a national population estimate. In consequence, and following Gill et al. (2001), the published national population index for each species was used (Pollitt et al. 2000). This is based on the Underhill Index (Underhill 1989) and provides a more accurate means of comparing changes in annual population numbers. Changes in the population total between successive years are calculated and the relative difference expressed as the index. In this case 1998/99 is used as the base year and the index set to 100.

Data analysis

For each species individually, the rates of population change for each site (henceforth termed simply 'rates of change') were calculated relative to the national population index. Thus, for each of the 19 species, a set of scatter plots (one for each site) was generated, with the local site population plotted against the national index for each year (e.g. Fig. 8.2). Each point on these plots represents one year. For each wetland, the rate of change was taken as the regression slope of the individual scatter plots.

Table 8.2: Results of the Spearman's rank (r_s) correlations between the initial population size (five-year mean) and the rate of population size change (relative to the national index) at each wetland site individually for the 19 waterbird species. r_s**R** = relative rates of population change, r_s**A** = absolute rates of population change. * = $P < 0.05$, ** = $P < 0.01$, *** = $P < 0.001$.

Species	r_sR	r_sA
Little grebe	0.175***	0.465***
Great crested grebe	-0.308***	0.332***
Cormorant	0.116**	0.620***
Mute swan	-0.345***	0.02
Bewick's swan	0.042	0.128
Whooper swan	-0.341***	-0.044
Canada goose	-0.221	-0.033
Dark-bellied brent goose	0.173*	0.719***
Shelduck	0.643***	0.329***
Wigeon	-0.448***	0.388***
Gadwall	-0.012	0.275**
Teal	-0.078	0.438***
Pintail	0.157*	0.018
Shoveler	-0.153	0.066
Tufted duck	0.009	0.151
Goldeneye	-0.126	0.316***
Red-breasted merganser	0.041	0.194*
Goosander	-0.185	0.072
Coot	-0.174	-0.028

Various regression models were tested (Minitab ™ version 13), however, in each case linear regression resulted in a significantly lower residual sum of squares term and was therefore considered the most suitable. For each wetland, the rates of change were calculated using both the raw numbers of birds (absolute rates) and the $\log_{10}$ local site numbers (relative rates).

To test for a buffer effect for each species, the relative rates of change in population size for each site were plotted against the initial population size at that site. Following Gill et al. (2001), the initial population size was calculated as the five-year mean population size (either peak counts or bird-days) for the years 1980/81-1984/85 for 15 species, from 1982/83-1986/87 for great crested grebe and coot, from 1985/86-1989/90 for little grebe, and from 1990/91-1994/95 for cormorant. One scatterplot was created for each of the 19 species, each point representing the rate of change at a site against the initial population size at that site (five-year mean). Finally, to determine the existence of a buffer effect, because the data for many of these plots appear to violate normality assumptions, Spearman's rank correlation was used. These analyses were repeated using absolute rates of change.

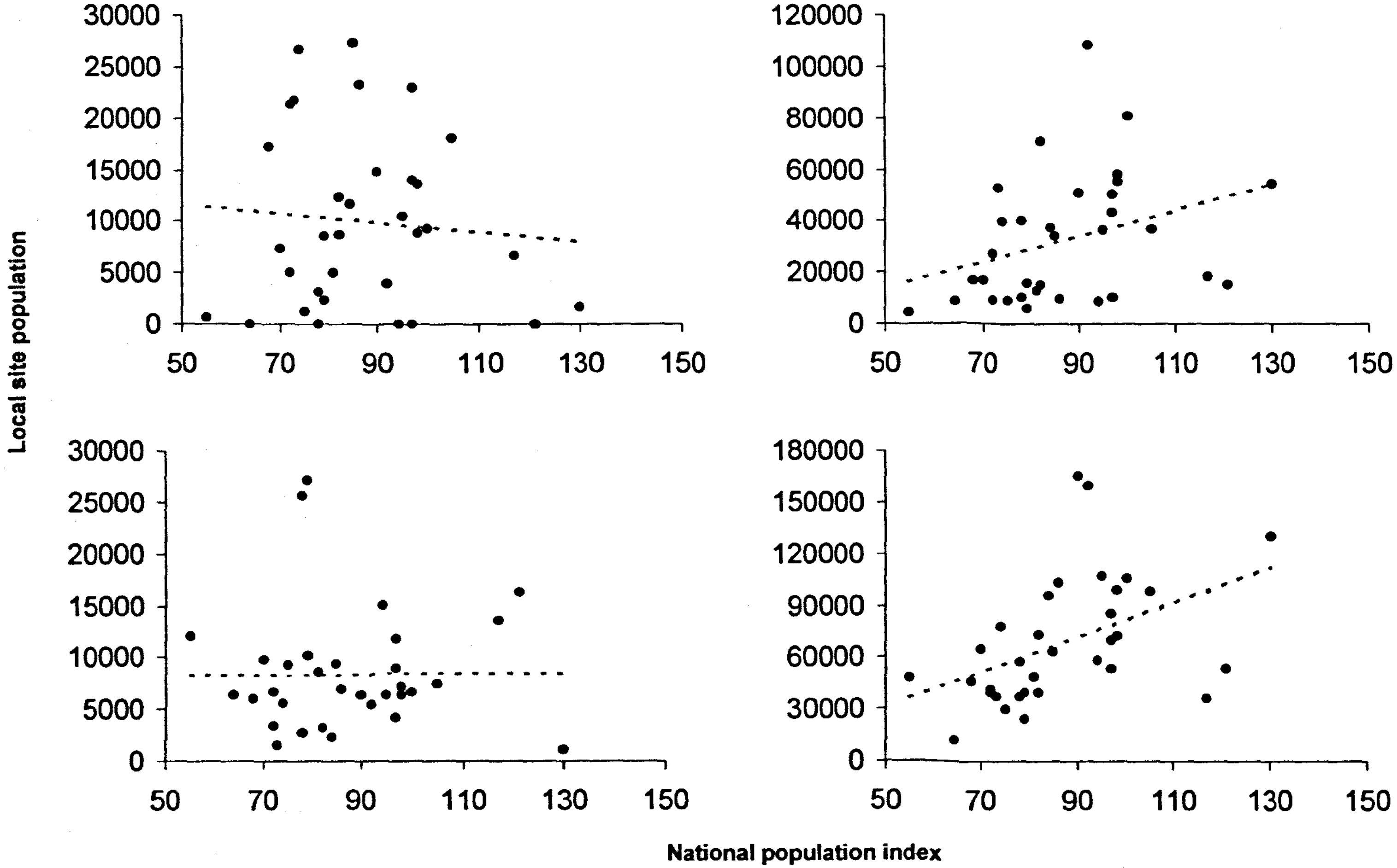

Figure 8.2: Local site populations plotted against the national population index in each year for four individual wetlands for goldeneye. Rates of population change for each site were taken as the slope of linear regression applied to each plot.

Additionally, to determine the degree to which each species has spread onto new sites as the national population increased plots were made of the national population index for each year against the number of previously unoccupied sites where a species was recorded in the same year. The strength of the relationship was assessed using Spearman's rank correlation. Previously unoccupied sites are defined as those for which either no individuals were recorded, or where numbers represent occasional visitors rather than colonisation events. More than two or three individuals recorded in the five preceding winter seasons distinguish occasional records from new colonisations.

8.3 Results

Despite increases in population sizes nationally (Pollitt et al. 2000), there was a significant negative correlation between initial population sizes and relative rates of change for only four of the 19 species analysed (great crested grebe, whooper swan, mute swan and wigeon) (Table 8.2).

Table 8.3: Results of Spearman's rank correlations between the national population index and the numbers of new sites occupied by a species in each year (1980/81-1998/99 unless otherwise stated in the Methods). *$P < 0.05$, **$P < 0.01$, ***$P < 0.001$.

Species	Rs
Little grebe	-0.332
Great crested grebe	-0.653**
Cormorant	-0.724**
Mute swan	-0.556**
Bewick's swan	0.245
Whooper swan	-0.379
Canada goose	-0.353
Dark-bellied brent goose	0.054
Shelduck	0.420
Wigeon	-0.493*
Gadwall	-0.394
Teal	0.153
Pintail	0.100
Shoveler	-0.344
Tufted duck	-0.308
Goldeneye	0.180
Red-breasted merganser	0.019
Goosander	-0.014
Coot	-0.307

The strongest negative correlation was for wigeon (Fig. 8.3a). For the remaining 15 species there was little evidence of a buffer effect and for shelduck there was a significant positive correlation (Fig. 8.3b). Seven species did show a slight negative correlation (Canada goose, gadwall, shoveler, teal, goldeneye, goosander and coot), however, for each this was non-significant and particularly low (<0.3), indicating little relationship between rates of change and initial population sizes (Table 8.2).

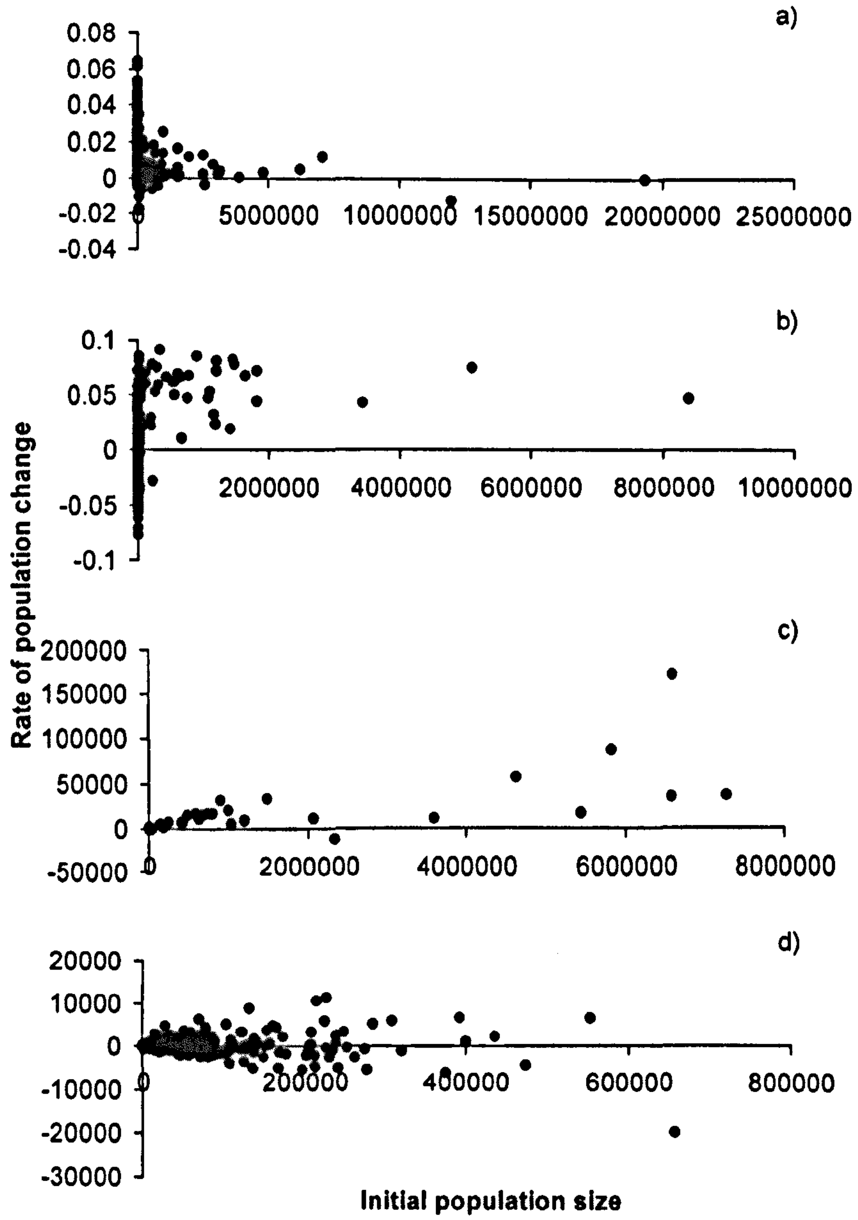

Figure 8.3: The initial population size (five-year mean in thousands) and the relative rate of population size change relative to the national population index at each individual wetland site for a) wigeon, b) shelduck, and the absolute rate of population size change for c) dark-bellied brent goose and d) Canada goose (calculated for the years 1980/81-1998/99). NB. The axes differ for each plot.

Considering correlations between absolute rates of change and initial population sizes, for 16 of the 19 species there was a positive correlation, with nine of the relationships being statistically significant (e.g. Fig. 8.3c; Table 8.2). This means that there were in fact greater additional numbers of birds in each year for those populations with larger initial numbers. For whooper swan, Canada goose and coot there was a slight negative trend (e.g. Fig. 8.3d), however, for each this was non-significant.

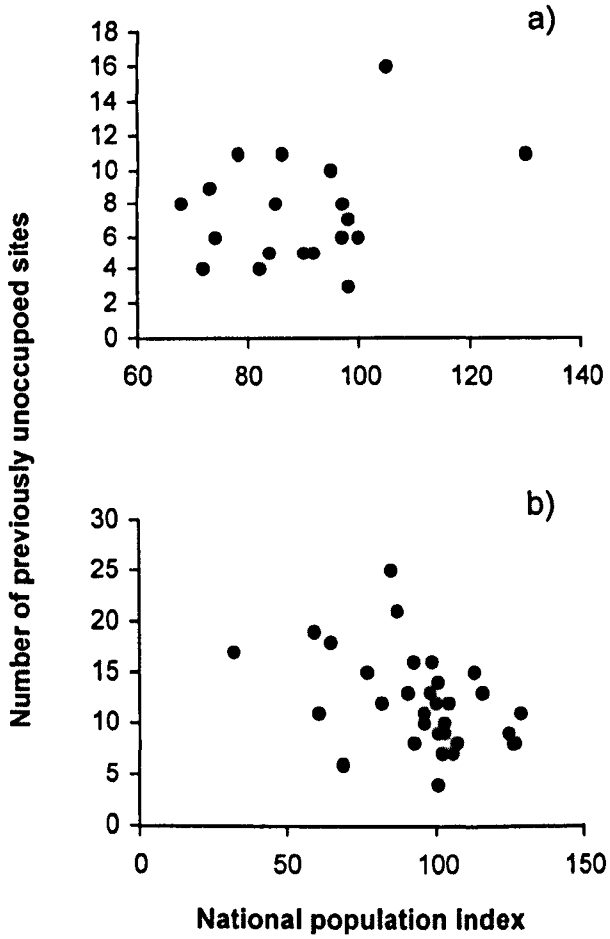

Figure 8.4: Correlations between the national population index for a species and the numbers of previously unoccupied sites where that species was recorded in each year (1980/1-1998/99) for a) goldeneye and b) wigeon.

The relationship between the numbers of additional (previously unoccupied) sites and the national population index was non-significant for 15 of the 19 species (Table 8.3). For seven of the 19 species the correlation coefficient was positive, with the numbers of sites increasing over time as the national index increased, however, for each this was non-significant (Fig. 8.4a). For four (great crested grebe, cormorant, mute swan and wigeon) of the 19 species there was a significant inverse relationship, where the numbers of additional sites decreased as the national population increased (Fig. 8.4b).

8.4 Discussion

These analyses provide little support for the general occurrence of a buffer effect amongst wildfowl and other waterbirds wintering on wetlands across Great Britain. Indeed, only four of the 19 species analysed showed a significant correlation between initial population sizes and relative rates of change (great crested grebe, whooper swan, mute swan and wigeon). In addition, despite annually increasing population sizes no significant expansion onto new sites has been observed for any of the 19 species analysed. On the contrary, the numbers of individuals are increasing on the same set of sites. Given that WeBS coverage is incomplete, concentrating on the larger, more important sites for waterbirds, any expansion onto the smaller, less desirable sites outside WeBS will not be evident from these analyses. Indeed, for those species where there was no significant relationship between initial population sizes and rates of change (either positive or negative), but where the rates of change for those sites with high initial population sizes were largely stable, it may be that individuals have been forced to relocate onto sites not included by WeBS (e.g. Bewick's swan, tufted duck and red-breasted merganser).

There are several reasons why these analyses may show no conclusive evidence of a buffer effect for the majority of species analysed. First, although these species are increasing in numbers over time, it may be that the preferred, high quality sites have not yet reached their carrying capacity. This means that, because the increases remain sustainable, there is no requirement for additional individuals to re-locate to alternative, lower quality sites.

Second, the relationship between initial population size and rates of change may be confounded by site area. Indeed, small sites in terms of available area will likely reach capacity relatively quickly and would, therefore, be expected to show a slower overall rate of change compared with the larger sites. Similarly, sites with larger available area for waterbirds have the capacity for sustained population increases for longer periods. Although it could be argued, therefore, that the testing of a buffer effect requires correction for area or analysis of densities rather than simply numbers, Brown's buffer effect concept states simply that there will be a negative correlation between the initial numbers at a site and subsequent rates of population increase (Brown 1969). Furthermore, other analyses at this scale using the same methodology do indeed report a negative relationship (e.g. Gill et al. 2001).

Third, the carrying capacity of a site need not exclusively be related to the population sizes of single species, possible exceptions being sites supporting large concentrations of, for example, European white-fronted goose or dark-bellied brent goose. Instead, the point at which a site is regarded by a single species as saturated may be determined also by the population sizes of other species wintering on that site. Given that several species share similar resource requirements then this is likely to some extent to be true of waterbirds. Testing for such an effect is complex, although considering the collective numbers of all WeBS species (see Pollitt et al. 2000 for details), there was a significant positive correlation between initial population sizes (all species summed) and the rates of change in total waterbird numbers ($r_s = 0.212$; $P < 0.0001$ (relative rates of change) $r_s = 0.312$; $P < 0.0001$ (absolute rates of change)). This suggests that, as yet, overall carrying capacity has not been reached for the majority of wintering sites.

Fourth, the existence of a buffer effect is conditional on the fact that the largest populations are to be found on the highest quality sites (Gill et al. 2001). However, population size is not an inevitable function of site quality. Indeed, observations made by (Menu et al. 2002) for snow geese in North America show no evidence of population growth regulation despite a dramatic increase in the North American population size. Rather, many sites have become severely impacted as populations have continued to increase above capacity, in many cases enough to compromise habitat integrity, although this happened because populations have suddenly been released from major mortality events as a result of hunting bans and it may be that the populations are no longer in equilibrium (Menu et al. 2002). Although some sites may be better for a species in terms of resource availability than others, site fidelity may be of overriding importance in determining which sites are the preferred wintering localities, particularly given the long life span of individuals of many of the species analysed. Evidence for site fidelity in waterbirds has been shown both for breeding sites (Blums et al. 2002) and wintering sites (Warkentin and Hernández 1996 and references therein). Additionally, conspecific attraction, where individuals are attracted to feeding/breeding sites by the presence of other individuals of the same species (Doligez et al. 1999, Brown et al. 2000) may influence a species' distribution across available sites. Thus, it may be that historical distributions have a stronger influence on current patterns as opposed to the present-day quality of the sites, particularly for those species showing high levels of winter site fidelity, such as certain geese and duck species, compared to ducks, which may show lower site fidelity in winter.

Finally, a high turnover of individuals at sites, such that individuals are using many different wetlands over the course of a winter season, may mask the expansion of individuals onto less suitable sites. However, using a bird-days approach based on a single count for each month will, to some extent, alleviate this problem.

Considering absolute rates of change, for 15 of the 19 species correlations between rates of change and initial population sizes were positive. Thus, for these species, those sites with larger initial population sizes are those gaining the greatest numbers of additional birds in each year. In terms of the current 1% threshold selection criterion for SPA classification these positive correlations, coupled with the lack of evidence for a buffer effect using relative rates of change amongst waterbird species at the national scale, is encouraging as it is likely that the favoured wetland sites for each of these species (i.e. those sites where a species preferentially locates in each winter) are still numerically the most important relative to potential 'spill-over' sites. These sites will, therefore, consistently be selected as potential Ramsar sites or SPAs following the current numerical site-selection guidelines. Nonetheless, should the national populations of these species continue to increase it is not certain whether this will continue to be the case, particularly if individuals are forced onto alternative, previously unoccupied sites once the preferred sites become saturated. In time, significant negative correlations between initial population sizes and the rate of change at a site may be expected, particularly for those species showing slight negative trends at present.

CHAPTER 9:

Population fluctuations and migratory waterbird conservation

9.1 Introduction

The persistence of species and habitats into the future is clearly of interest to planners, managers and conservationists alike (Rodrigues et al. 2000b). Not only is it vital in light of increasing anthropogenic pressure on the remaining land (which is likely to diminish the opportunities for conservation in the future), but also it will enable planners and managers to confidently justify a site's inclusion in a protected area network should conflicts arise. In this respect, the evaluation of the effectiveness of such networks is not only essential to assist managers in their work, but to promote accountability. Indeed, notwithstanding the wealth of literature designed specifically to optimise the site-selection process (see for example Bedward et al. 1992, Pressey et al. 1993, Church et al. 1996, Margules and Pressey 2000, Possingham et al. 2000, Cabeza and Moilanen 2001) those painstakingly selected sites must be maintained in such a way that, over time, the populations/habitats are better protected than the surrounding countryside. According to the IUCN, an equal amount of effort should be invested in the on-going management of protected areas as in selecting them in the first place (Hocklings et al. 2000). In reality, however, the management of protected areas is dictated by social, political and economic factors and is consequently often confined to the level of the individual site rather than considering the national network as a whole. Nonetheless, regulatory bodies are increasingly demanding information on management effectiveness to assess whether results are compatible with the level of funding and resources provided (Hocklings et al. 2000). Additionally, many such agencies are obliged to assess whether achievements at the national level are in line with policy objectives. For example, signatories to the Convention on Biodiversity must present to the Conference of the Parties reports on measures taken to implement the various Articles of the Convention and their effectiveness (Article 26).

In addition to the numerical guidelines for the selection of SPAs and Ramsar Sites (discussed in detail in Chapter 2), the JNCC criteria stipulate that selected sites should 'contribute significantly to a species' population viability both locally and as a whole'. The specific attributes thought to contribute most to population viability include 'small-scale population fluctuations around a stable population size'. In effect, this means that for individual SPAs there is a lower limit on the numbers of individuals of each target species present on that site. To successfully maintain numbers above these baseline levels site managers can, therefore, only realistically take measures that either increase the mean numbers of individuals, or decrease the variance in these numbers over time. These two management options are, however, not mutually exclusive. Indeed, by attempting to maintain mean numbers above initial levels it is likely that there will also be a reduction in the variance over time (a logical step, for other reasons, given that all else being equal, populations with higher variance are at a greater risk of local extinction; Pimm 1991).

If these EU and Ramsar Convention conservation requirements and management actions are indeed being addressed effectively at both the local and national management level in Great Britain, then the expectation is that populations of individual species on sites managed as part of the PS network would show a lower variance in numbers of individuals over time compared with populations of the same species on other non-managed (under either the EU Birds Directive or the Ramsar Convention) wetland sites. Furthermore, these actively managed and protected sites should retain greater total numbers of birds over time, particularly given that signatories to the Birds Directive must submit a report every three years detailing overall progress and achievements (Article 12). Similarly, Contracting Parties to the Ramsar Convention on Wetlands of International Importance are required to report on progress in implementing the responsibilities of the Convention at the triennial Conference of the Parties. However, unlike the Birds Directive there is no legal requirement to do so. To test these assumptions, 21 selected waterbird species on PS and non-PS wetlands in Great Britain were compared in relation to their mean numbers of individuals and the associated variance in numbers over a six-year period (1993/94-1998/99). Based on these comparisons, I provide suggestions for future site-selection and management considerations.

9.2 Methods

Data

WeBS data for the years 1993/94 to 1998/99 inclusive (representing six winter seasons) were included for a selection of 21 waterbird species wintering in Great Britain for which WeBS provides good annual coverage. This includes, little grebe, great crested grebe, cormorant, Bewick's swan, whooper swan, European white-fronted goose, dark-bellied brent goose, shelduck, wigeon, gadwall, teal, mallard, pintail, shoveler, pochard, tufted duck, goldeneye, smew, red-breasted merganser, goosander and coot. Additionally, the summed numbers of all waterbird species (excluding waders) counted by WeBS for each wetland site will be referred to as 'all species'. This six year period was chosen to maximise the power of the statistical tests used throughout this Chapter.

The PS network is not static and additional sites are classified in each year (see Fig. 2.1). To allow for accurate comparison, the PS network was, therefore, taken as the 79 sites classified as SPAs or Ramsar Sites prior to and including the 1993/94 winter season. This excluded any wetland classified/designated as an SPA/Ramsar Site from 1993/94 onwards. In this way, there were sufficient numbers of sites included in the analysis and the numbers of PS remained constant for all years. The numbers of PS and non-PS wetlands where each of the 21 species were recorded between 1993/94 and 1998/99 are shown in Table 9.1. Similarly the numbers of non-PS network sites were also kept constant in all years of the analysis, although the numbers varied for each of the 21 species analysed (see Table 9.1).

Table 9.1: Numbers of non-PS and PS wetlands included in this analysis where each of the 21 species was recorded as present for the years (1993/94 to 1998/99).

Species	Non-PS	PS
Little grebe	1245	60
Great crested grebe	898	58
Cormorant	1229	54
Bewick's swan	104	22
Whooper swan	393	43
European white-fronted goose	132	33
Dark-bellied brent goose	88	27
Shelduck	533	41
Wigeon	1148	51
Gadwall	673	51
Teal	1238	61
Mallard	1724	68
Pintail	449	43
Shoveler	733	55
Pochard	1216	63
Tufted duck	1429	67
Goldeneye	1149	71
Smew	260	41
Red-breasted merganser	356	51
Goosander	999	56
Coot	1351	68

Analyses

To compare the existing wetland PS network with non-PS wetlands, peak annual counts for each of the 21 species and for all WeBS species summed were calculated for each site individually. For each year (1993/94-1998/99), in line with the methodology adopted by WeBS (see Pollitt et al. 2000), these site counts were taken as the maximum of the individual monthly counts from September to March.

For each of the 21 species, the mean and variance in numbers were calculated for each individual site over the six-year period. These values were then $\log_{10}$ transformed for analysis. To examine the effects of temporal trends in these site counts between 1993/94 and 1998/99, all analyses were conducted both using raw and detrended abundance data (detrended using a quadratic trend model; Minitab™ Version 11.12). However, because the results were largely identical only the results of the analyses using detrended data are reported here. To account for the mean dependence of the variance in numbers of individuals, all mean-variance comparisons were made using Analysis of Covariance (ANCOVA), with $\log_{10}$ site means as the covariate, the $\log_{10}$ variance in numbers as the dependent variable, and site type (PS or non-PS) as the fixed factor.

9.3 Results

Contrasting PS and non-PS wetland totals

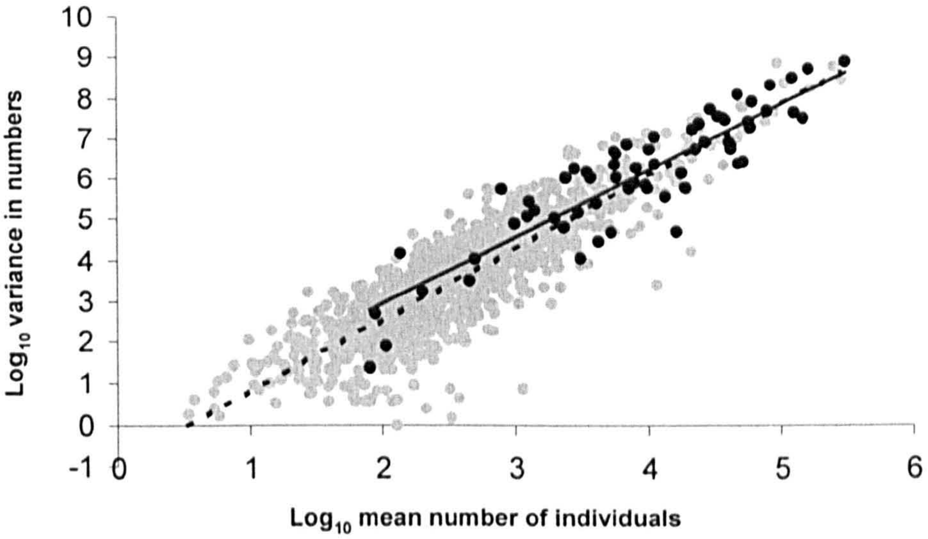

Figure 9.1: The $\log_{10}$ variance in numbers and the mean numbers of individuals for sites classified/designated as PS (black circles) and for non-PS (grey circles) calculated between 1993/94 and 1998/99 for all waterbird species included by the Wetland Birds Survey. The lines (fit of the variance to mean power function or Taylor's law) represent the linear trends for PS (solid line) and non-PS (dashed line) wetlands.

Considering the total numbers of all WeBS species (see Pollitt et al. 2000 for details), despite only 38% of the PS wetlands being ranked within the top fifty sites for numbers of individuals, there was a significant difference between the mean numbers of individuals (1993/94-1998/99) recorded on the PS network compared with non-PS wetlands ($Z = -10.275$; N

= 1137; $P < 0.0001$) (Fig. 9.1). Specifically, the general trend was towards greater numbers on the PS network, with a mean (±SD) of 32,455.17 (±52,901.98) birds, compared to 2,572.03 (±14,870.60) birds for the non-PS wetlands. Despite a greater range of site means for the PS network, the numbers of individuals on these sites tended towards the larger mean values (Fig. 9.2).

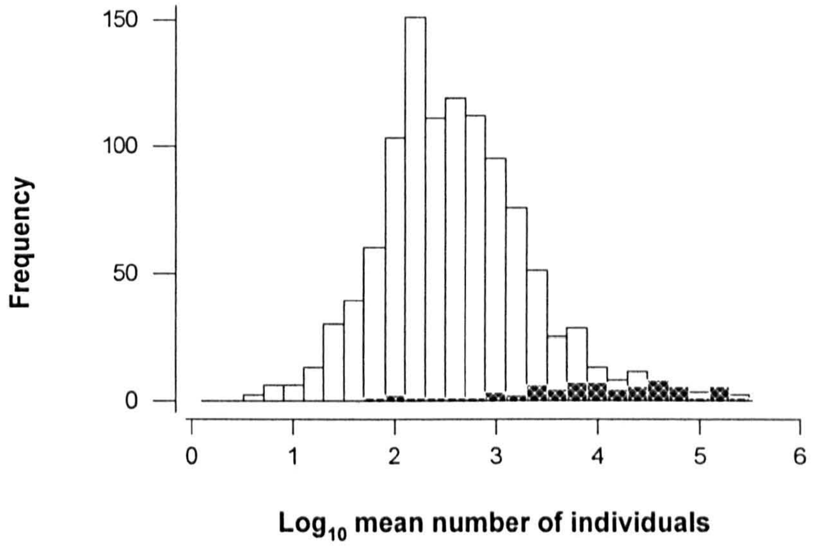

Figure 9.2: Histogram showing the frequency distribution of the $\log_{10}$ mean numbers of individual waterbirds calculated for each individual PS (filled, black bars) and non-PS wetland (white bars). The cross-hatched bars indicate the overlap between the two site types.

Considering each of the 21 species individually, for 18 species less than half the PS were ranked within the top fifty sites according to the number of individuals of that species supported (sites ranked separately according to the mean numbers of individuals for each of the 21 species). For the remaining three species (Bewick's swan, European white-fronted goose and dark-bellied brent goose) more than half the PS were in the top fifty sites. In addition, for 17 of the 21 species, the top site ranked for mean numbers of individuals was not only a PS, but was also listed as nationally important for that species (Listed site). Conversely, for goldeneye, smew, goosander and red-breasted merganser, the top site was not included in the PS network.

For each species, the percentage of PS in the top fifty sites for site numbers correlated positively with a species' aggregative tendency, taken as the percentage of the national total found in the ten most populous sites ($r = 0.536$; *d.f.* = 21; $P < 0.05$) (Fig. 9.3). Specifically, species that tend to form dense flocks, such as for example European white-fronted goose and Bewick's swan, generally had a greater percentage of the PS within the top fifty sites for that species (see also Chapters 2 and 3). In contrast, species with a more dispersed distribution pattern, such as tufted duck and goldeneye, tended to have a significantly lower percentage of their PS ranked within the top fifty sites for site numbers.

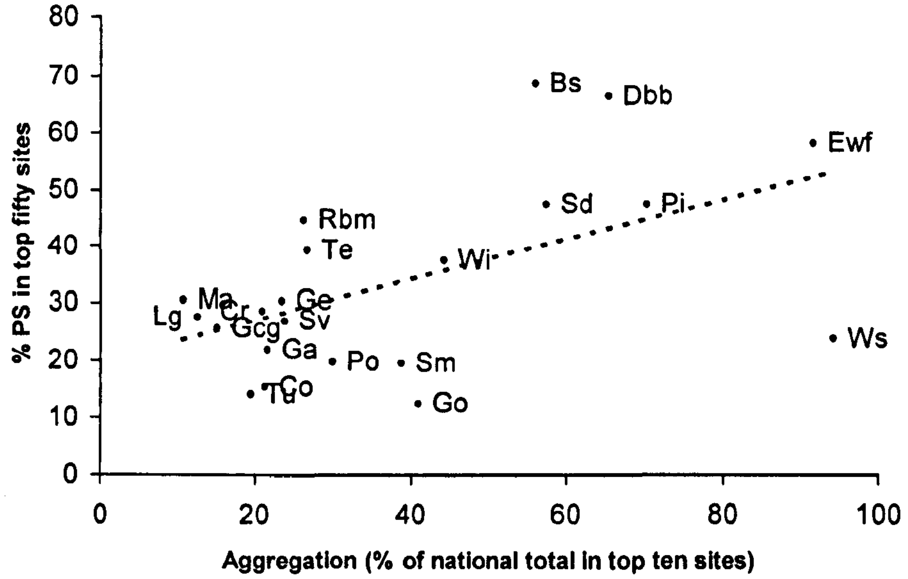

Figure 9.3: The % of PS in the top fifty sites for mean numbers of individuals and the degree of aggregation (% national total within the top ten most populous sites for a species) for: Bs) Bewick's swan, Ws) whooper swan, Ewf) European white fronted goose, Dbb) dark-bellied brent goose, Sd) shelduck, Wi) wigeon, Ga) gadwall, Te) teal, Ma) mallard, Pi) pintail, Sv) shoveler, Po) pochard, Tu) tufted duck, Ge) goldeneye, Sm) smew, Rbm) red-breasted merganser, and Go) goosander.

For 19 of the 21 species the frequency distributions of the mean numbers per site were significantly different between the PS network and non-PS wetlands. In general, for the PS network the distribution of sites tended towards the larger mean numbers of individuals, whereas there was a more even distribution for the non-PS wetlands (e.g. Pintail, Fig. 9.4a). Conversely, for whooper swan (Fig. 9.4b) and goosander, there were no significant differences between the frequency distributions of the PS and non-PS wetlands.

Mean-variance comparisons for the PS and non-PS wetlands

For all WeBS species and each of the 21 species individually there was a strongly significant positive correlation between the mean numbers of individuals and the variance in numbers, under logarithmic transformation, for both the PS and the non-PS (Fig. 9.1 and Fig. 9.5). Indeed, for each of these plots, r^2 values were high (0.68 to 0.98 for PS wetlands and 0.76 to 0.95 for non-PS). In other words, the variance to mean power function/Taylor's power law

($\sigma^2 = a\mu^b$ where σ^2 is the variance in numbers of individuals over time; μ, the mean number of individuals; and a, the intercept and b the slope of the log mean-log variance regression line) fitted the observations well (Taylor 1961, Maurer and Taper 2002).

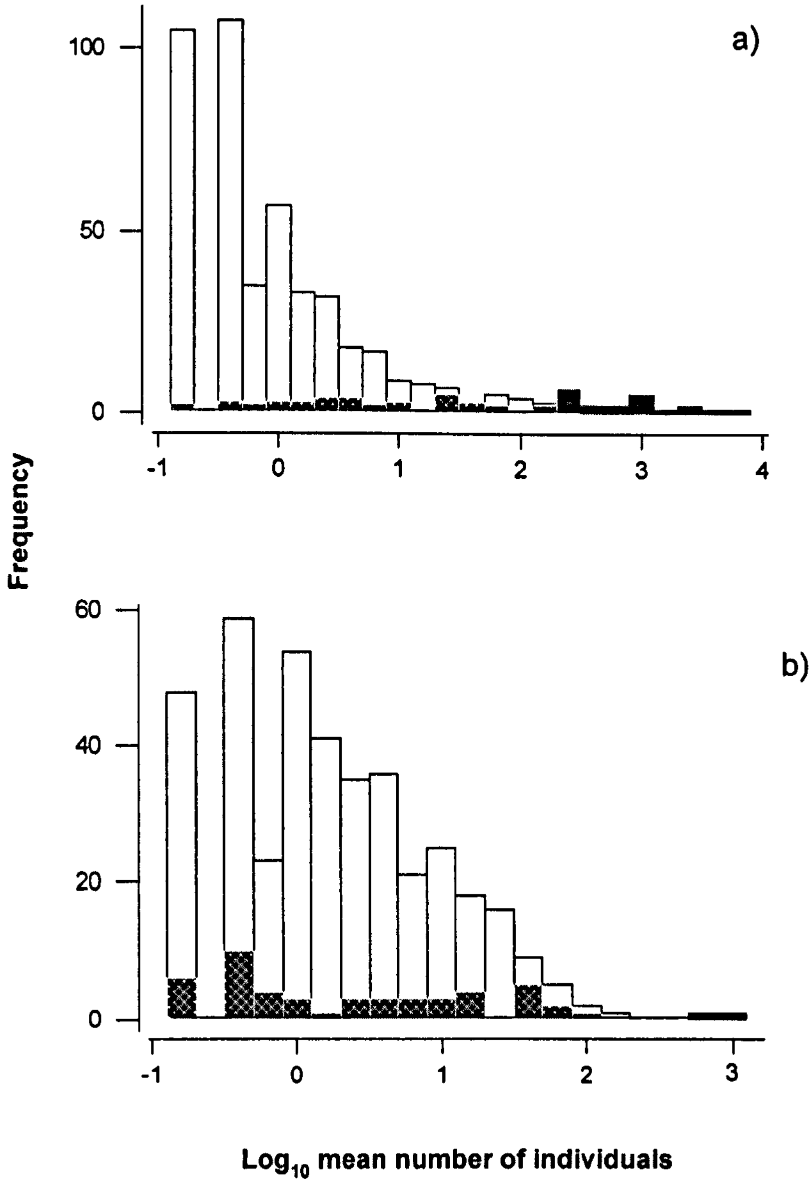

Figure 9.4: Histogram showing the frequency distribution of the $\log_{10}$ mean numbers of individuals for PS wetlands (filled, black bars) and non-PS wetlands (white bars) for a) pintail and b) whooper swan. The cross-hatched bars indicate the overlap between the two site types.

Table 9.1: The *F* test for equality of slopes and intercept comparisons for each of the 21 species. The analysis was carried out using $\log_{10}$ detrended abundance data within a standard Analysis of Covariance (ANCOVA). * $P < 0.05$. Significant differences in the mean-variance slopes (*F* test) exclude these species from further analysis.

Species	Equality of slopes	Intercept comparisons
Little grebe	12.760*	
Little grebe	4.470	3.902
Great crested grebe	20.607*	
Cormorant	0.133	2.054
Bewick's swan	13.210*	
Whooper swan	0.024	1.324
European white-fronted goose	21.559*	
Dark-bellied brent goose	2.271	5.519
Shelduck	9.197	1.863
Wigeon	0.001	4.663*
Gadwall	6.038	0.559
Teal	0.02	6.989
Mallard	1.962	0.089
Pintail	0.343	0.114
Shoveler	0.112	0.639
Pochard	2.996	0.407
Tufted duck	1.038	4.488
Goldeneye	7.302	0.469
Smew	0.049	0.138
Red-breasted merganser	0.345	0.398
Goosander	0.282	0.215
Coot	2.908	2.388

Considering all WeBS species together, *F* test comparisons showed significant differences in the slopes of the mean-variance relationships between PS and non-PS wetlands (Table 9.2). Specifically, where the $\log_{10}$ mean number of individuals was low to intermediate, PS wetlands showed a greater associated variance in numbers compared with non-PS (Fig. 9.1), although the difference between the two slopes decreased as the mean increased.

Conversely, at high $\log_{10}$ mean population sizes, the reverse was true; the non-PS showed a greater variance in numbers of individuals for a given $\log_{10}$ mean population size (Fig. 9.1). In addition, the mean-variance slopes for great crested grebe, Bewick's swan and European white-fronted goose were significantly different ($P < 0.001$), which also excluded these species from the ANCOVA analysis.

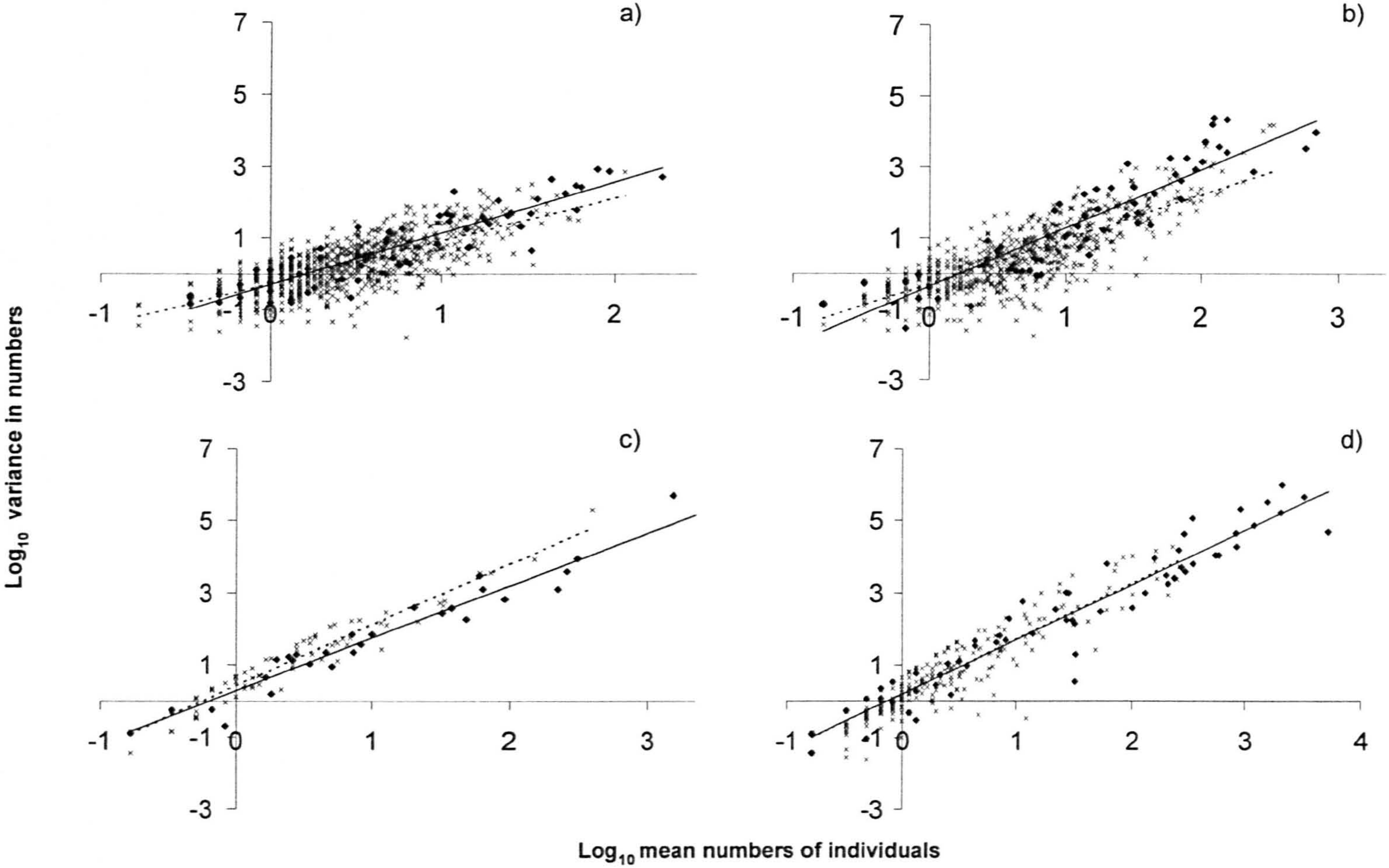

Figure 9.5: The relationships between the log$_{10}$ variance in numbers and the log$_{10}$ mean number of individuals (fit of the variance to mean power function or Taylor's law) for sites classified/designated as PS (black diamonds) and for non-PS (crosses) calculated between 1993/94 and 1998/99 for: a) little grebe, b) great crested grebe, c) Bewick's swan and d) pintail. The lines represent the linear trends for PS (solid line) and non-PS (dashed line).

Nevertheless, for great crested grebe and Bewick's swan the variance in numbers for a given mean number of individuals was generally greater for the PS than for the non-PS. The reverse was true, however, for European white-fronted goose.

Considering only the 18 species for which there were no significant differences in the slope of the mean-variance relationship for PS and non-PS wetlands (Table 9.2), intercept comparisons (variance in numbers) between PS and non-PS wetlands were not statistically significant for 17 of the 18 species. Only comparisons for wigeon showed significant differences, although this was borderline (Table 9.2). Moreover, for this species, for a given mean population size, the variance in numbers was greater for the PS wetlands compared with non-PS wetlands.

9.4 Discussion

The need for appropriate management strategies for protected area networks is well known. Although, considering all WeBS species, it could be argued that, for those PS with the largest numbers of individuals, management may be working, the results of this analysis demonstrate a clear distinction between the ability to influence the numbers of birds wintering at a particular site in any given year, and an overall inability to influence the annual variation in numbers. Specifically, by applying the appropriate management efforts, for example simply by increasing the area of available habitat, a greater proportion of the annual national total of a particular species could be attracted to a single site in a given year (Chapter 3.1). There are, however, several reasons why it should be possible to increase numbers at a given site.

First, in view of the current means by which important sites for waterbirds are selected in Great Britain, the significantly greater mean numbers of birds supported by the PS as opposed to non-PS wetlands could reflect the fact that these sites were simply better to begin with (see Chapter 3.2). Thus, management activities on the PS may not, in fact, be having any effect on the populations of waterbirds that return in each winter season. In this respect, comparisons between the rates of population change over time (rates calculated as the slope of a linear regression between the $\log_{10}$ numbers of all WeBS species at a site and the national population total for the years 1994/95-1998/99) showed no significant difference between PS and non-PS wetlands.

Second, waterbirds have been shown to exhibit high levels of site fidelity, both to breeding (Blums et al. 2002) and wintering grounds (Warkentin and Hernández 1996 and references therein; Chapter 2), therefore, historically favoured sites may have remained so irrespective of site management. However, the significantly greater mean numbers of individuals recorded on PS rather than non-PS wetlands implies that, at the very least, management of these sites at the local site level has not adversely affected these important wetland sites and has been successful at maintaining populations at levels similar to those observed pre-classification.

Despite the significant differences in the mean numbers of individuals between PS and non-PS wetlands for each species individually, considering all WeBS species together, less than 40% of the top fifty sites ranked in order of mean numbers of individuals were included in the PS network. Nonetheless, these PS tended towards larger numbers of birds when compared with the non-PS wetlands. Although this was expected given that many of these sites were deliberately selected on the basis of their total waterbird numbers (i.e. total numbers of all WeBS species; Stage one of the SPA selection guidelines in Great Britain), it is highly encouraging, particularly as many of these wetland SPAs have been under active management since the early 1980s. In addition, not all species on a particular site are target species listed in the EU Birds Directive. This would suggest that the classification/designation of wetland sites as SPAs/Ramsar Sites is often beneficial not only to focal species (those species for which the site was originally classified) but also to other Annex I/II species present on the site.

In contrast to the apparent success of local site management in holding or attracting greater mean numbers of wintering waterbirds to PS as opposed to non-PS wetlands, it would appear that site-based management activities have not imparted a greater level of stability to the overall numbers of waterbirds returning each winter. In the absence of PS management, the numbers of individuals on many sites appear, in fact, to be more stable. However, this may simply reflect the fact that these sites were more stable in the first place rather than ineffective management. There are several possible reasons for the lack of significant differences between the two site types with regards to the variance in numbers.

First, it is possible that inadequacies in the data set are masking the real patterns (see Chapter 1). However, although it is recognised that errors in these data inevitably exist, it is unlikely that these alone will be sufficient to produce lack of significant differences between the two site types with regards to the variance in numbers. Indeed, it remains more likely that the practicalities of implementing the various requirements of the Birds Directive are directly responsible for the lack of significant difference between PS and non-PS wetlands.

Second, assuming that local site managers are aware of national and international requirements for management, conflicts of interest are inevitable at sites with more than one official designation (such as Sites of Special Scientific Interest (SSSI), National Nature Reserve (NNR), Special Area of Conservation (SAC)) or where more than one species is listed as a management priority under the SPA selection guidelines (see also Chapter 7). For example, activities aimed at the conservation of one species or assemblage could potentially negatively impact another, such as provisions for diving versus dabbling waterbird species or improving grasslands for waders versus geese (Vickery et al. 1997). Therefore, although site managers are expected to target the populations of all species listed as nationally important at a particular site, conflicting management requirements may mean that it is virtually impossible to provide for all such species on any single wetland (see Chapter 7). Notable exceptions are the largest sites, such as The Wash, where there is a sufficiently large area to incorporate a variety of management options simultaneously.

Third, it may simply be that the measures undertaken to protect and maintain waterbirds are not having the desired effect, either because not enough is currently known about what is actually required, or that the intensity of such activities is insufficient to produce a discernible effect, despite the best efforts of the site managers.

Fourth, rather than being a case of inappropriate management or difficulties with the application of the Directive itself, the lack of significant differences in population variance in numbers between PS and non-PS wetlands may simply reflect the fact that the management of migratory species is not easily confined to isolated sites (Storrs and Finlayson 1997, Esler 2000, Turner et al. 2000). Migratory waterbird populations are not static in their wintering areas, with individual birds capable of moving great distances, in many cases greater than 1000km. In consequence, in any one year, the distribution of individuals is determined not only by the characteristics of the local site, but also by large-scale environmental processes (such as the breeding conditions and weather conditions along migration routes; Gill et al. 2001, Goss-Custard et al. 2002, Webster et al. 2002).

Finally, in addition to these international scale influences, within a single season there is evidence to suggest that for many species, individuals move between multiple sites (Moser 1987, Goss-Custard and Durrel 1990, Skagen and Knopf 1993, Rehfisch et al. 1996, Farmer and Parent 1997, Pradel et al. 1997, Madsen et al. 1998), with more than one wetland being critically important in a particular year. In consequence, to successfully protect and manage target populations within protected areas it is essential to understand the spatial scale over which population processes occur and to respond accordingly in management plans. In this respect, management of PS in Great Britain will require intervention both at the local scale, so as to maintain the internal dynamics of the system, and also at a wider scale, to incorporate factors external to the local site but which nevertheless influence local site dynamics (Knight 1998). However, this is inevitably a mammoth undertaking, requiring conservation planning at scales previously not widely considered (Opdam et al. 1995, Suter 1998), and will inevitably cross both geographical and political boundaries. One example of such a programme is the North American Waterfowl Management Plan (NAWMP) signed by Canada, the United States and Mexico (Myers et al. 1996). Nevertheless, even the ultimate success of such a remarkable, multi-national effort will depend on the recognition of the importance of both local and large-scale processes on local site populations (Shea and the NCEAS Working Group On Population Management 1998, Slocombe 1998, Maurer and Taper 2002, Webster et al. 2002).

9.5 Conclusions

In practice, local site management needs to concentrate on ensuring that wetland sites remain suitable for these species, to attract an increasing proportion of the winter population away from non-protected and non-PS wetlands. In this way a greater level of protection can be afforded to all focal species (Annex I and Annex II species listed in the Birds Directive) and, despite continued and progressive alteration of the natural environment, these species will be better able to persist into the future.

CHAPTER 10

'If conservation biology is ineffective in helping to stop something as globally significant as the devastation of the Indonesian rainforests, then what, please, is the point of it?' (Whitten et al. 2001)

The gap between conservation theory and conservation practice has been maintained over the past few decades and, despite the best efforts of all concerned, this enduring polarity is likely to have undesirable consequences for the persistence of biodiversity and the success of conservation efforts. The purpose of this last chapter is, therefore, not to discuss the implications of each of the preceding chapters individually as this has already been largely done in the associated concluding sections. Rather, the intention is to address the probable reasons for the lack of communication and coordination between theoreticians and practitioners within the field of conservation biology, and to provide suggestions as to how to overcome these difficulties, particularly with regards to the conservation of waterbirds in Great Britain. Finally, this thesis will end with a discussion of the potential and required role of conservation biology and conservation biologists now and into the future.

Please mind the gap:
Conservation theory versus conservation practice

10.1 Introduction

Over the past three decades considerable effort has been directed towards devising, testing and adapting the theoretical principles of conservation biology (Pullin and Knight 2001). Indeed, conservation scientists have developed various increasingly sophisticated tools to aid in the selection and identification of priority areas (e.g. Chapters 4-7). Nonetheless, conservation practitioners (individuals and institutions making conservation related decisions) frequently act without the benefit of this knowledge (Adams et al. 2002). In consequence, the impact of conservation theory on practical conservation issues has remained disturbingly small. For example, the identification of priority wetlands for wintering waterbirds in Great Britain is carried out using a more traditional site selection approach, based on the attainment of numerical thresholds for population size (discussed in detail in Chapter 2). In an attempt to draw attention to and ultimately to eliminate this lack of cooperation/coordination, Prendergast et al. (1999) called for closer dialogue between theoreticians and practitioners of conservation biology. Five years on, however, it would appear that their advice has been largely disregarded. Indeed, there remain disturbingly few examples of the practical implementation of conservation theory to real world conservation problems (see Table 10.1 for notable exceptions). As an indication of the current extent of this problem, only a small percentage (<10% in each case) of the abstracts for presentations to the 2002 meeting of the Society for Conservation Biology (SCB) were concerned with either implementation (the application of results to management) or the evaluation of activities (Kleiman 2003). Following on from this, a similar analysis of the presentations (posters and talks; N = 141) to the 2002 and 2003 Student Conferences on Conservation Science (SCCS), confirmed the overall trend towards problem description (31.9% of the total abstracts) and basic biological research (34.0% of abstracts; Fig. 10.1). By contrast, none were concerned with the application of results to management and only 2.8% ($N = 3$) with the evaluation of activities (Fig. 10.1). Although, as Kleiman (2003) pointed out, an analysis of abstracts may not accurately reflect the actual content of these presentations, these results nevertheless represent a disturbing trend within the field of conservation biology.

Despite the evident bias towards describing conservation problems and basic biological surveying, there are some notable examples of the application of conservation theory to real world planning issues, although, it should be noted that none of these relate to the selection of priority areas for wintering waterbirds. For example, Cowling (1999), Cowling and Pressey

(2003) and Cowling et al. (2003a) provide an overview of the conservation plan for the succulent karoo hotspot of South Africa's Cape Floristic Region (CFR). The project, commissioned by South African National Parks (SANP), although still in the planning phase, aims to identify the highest conservation priorities to guide the allocation of available funds in the area. Specifically, conservation targets were set for 851 Red Data Book species and sub-species relating to the overall area and numbers of localities required for each species individually. To select the minimum set necessary to represent each of these threatened species at least once, a greedy heuristic algorithm (Pressey et al. 1993) was employed both including and excluding the existing reserves. Thus far, work on several priority reserves identified using this procedure is well under way, although, the required land has yet to be purchased (Table 10.1).

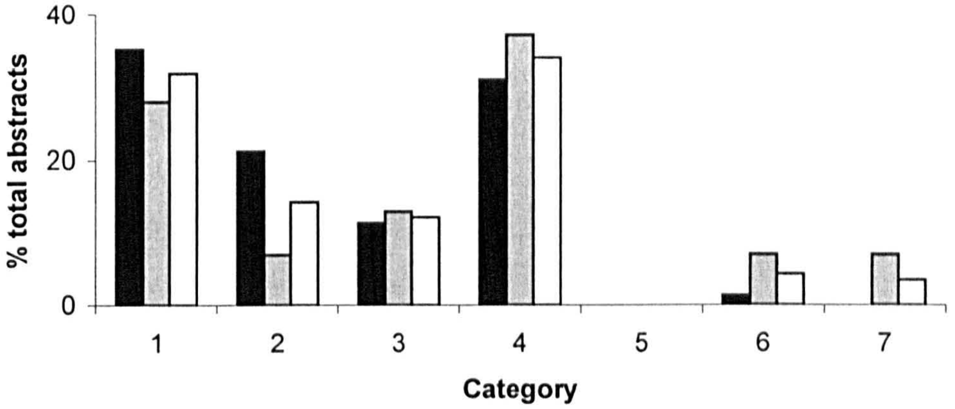

Figure 10.1: The % of the total numbers of abstracts (*N* = 141) for talks (grey bars), posters (black bars) and both talks and posters together (white bars) for the 2002 and the 2003 Student Conference on Conservation Science in each of seven categories. Each abstract was placed into one of the categories: 1) description and definition of a problem, often through statistical analysis of data; 2) development of a model; 3) directed research to find solutions; 4) basic biological monitoring (e.g. descriptive surveys, monitoring and basic research); 5) implementation (application of results to management); 6) evaluation of activities; and 7) none of the above.

A second example, is the successful application of a systematic conservation planning approach to a regional planning exercise in the Columbia Plateau ecoregion of the northwest United States (Davis et al. 1999). Designed and carried out by The Nature Conservancy (TNC), the aim of this project was to design and test a prototype planning process to be used to identify conservation priorities within the other 62 ecoregions of the United States. The proposed network of priority sites was selected using a linear integer programming model (the Biodiversity Management Area Selection model; Table 10.1).

Table 10.1: Examples of the practical application of conservation theory to conservation planning.

Author	Region	Features	Objective	Method of site selection	Implementation
Lombard et al. (1997)	The Agulhas Plain, Cape Floristic Region, South Africa	Fynbos communities including 11 vegetation types and 99 endemic species	To identify a minimum set to represent all species and vegetation types using a range of targets for coverage.	Stepwise heuristic incorporating an adjacency constraint	Partially implemented. The Agulhas National Park was officially declared in 1999 and was approximately half the area suggested in the study.
RACAC (1996)	Eastern New South Wales	Forest types	To identify priority sites for conservation in the region to complement existing reserves.	Greedy heuristic (C-plan)	The planning process was carried out in response to the request for new areas. Since then, nine new parks and nature reserves, extensive new wilderness areas and agreements on the supply of hard wood have been set up.
Cowling et al. (2003a)	Cape Floristic Region, South Africa	Surrogates for ecological and evolutionary processes, Proteaceae, vertebrates, mammals	To produce a conservation plan that achieved conservation targets for biodiversity pattern and process.	Iterative heuristics Delphi analysis	The aim is for the plan to be implemented over the next 20 years.

Richardson and Funk (1999)	Guyana	Ten taxonomic groups of plants and animals	To design a system of reserves that would protect biodiversity but also respecting the rights and land use of Amerindians, while at the same time allowing economic development.	Iterative heuristic (C-Plan)	The government has agreed to consider just two areas as the foundation of the National Protected Area System.
Cowling (1999)	Succulent Karoo desert, South Africa	Red Data Book flora	To design a representative system of reserves for the Cape Floristic Region, subject to budgetary constraints.	Minimum set using iterative heuristics	South African National Parks has committed itself to the process and work is far advanced.
Davis et al. (1999)	Columbia Plateau ecoregion, USA	Vegetation types and rare/threatened species	Identifying priorities for species representation while balancing the dual objectives of efficiency and site suitability.	Linear integer programming model (Biodiversity Management Area Selection model) and expert knowledge	The planning process was implemented by The Nature Conservancy.
Faith et al. (2001a, 2001b, 2001c, 2001d)	Papua New Guinea	87 plant and animal taxa	To establish a national protected area network that also accounts for timber production.	Iterative heuristics (TARGET software)	The process is on going and part of the development of a national biodiversity conservation plan.

Pressey et al. (2000)	New South Wales	Vegetation types	To achieve a fully representative set of reserves and other conservation areas for terrestrial and marine environments.	Gap Analysis	The process is on going and part of the development of a national biodiversity conservation plan.
Howard et al. (1997)	Uganda	Woody plants, small mammals, birds, butterflies and large moths	To manage 50% of the forest estate primarily for environmental protection, with 20% designated as Strict Nature Reserves (SNRs).	Scoring/ranking using diversity, richness and rarity	The planning and analyses took 5 years and cost more than $US1 million and is part of an on-going forest-management programme on behalf of the Ugandan government.

10.2 Problems, conflicts and solutions

There are many reasons for the enduring gulf between conservation theory and practice, which can be subdivided into five general categories, although none are mutually exclusive:

1. Lack of cooperation/coordination between and within research groups and practitioners.
2. Mistrust of algorithms and new technology.
3. Lack of knowledge/awareness.
4. Circular arguments/lack of consensus.
5. Stakeholder involvement/social agenda.

I will review each of these factors in turn and provide suggestions as to how they can be overcome in order to facilitate the translation of effective and efficient conservation planning theory to real world conservation problems, with particular reference to waterbirds in Great Britain.

Lack of cooperation

Problems

Despite broad agreement concerning the fundamental principles of conservation biology and of global conservation goals, working on the details has more often than not resulted in conflict within the conservation movement and divided conservationists to the detriment of what they are trying to achieve (McShane 2003). In reality very little collaboration between researchers and practitioners and also between different organisations and institutions actually takes place (Prendergast et al. 1999). Rather, managers criticise researchers for failing to address relevant questions for application to real world conservation efforts in favour of wasting money on more esoteric issues (i.e. the development/refinement of priority site selection algorithms). In particular, practitioners would argue that the ecologist's view is far removed from the pragmatic approach of the practitioner who operates in the world of budgets, targets set out in action plans, and responsive action to avert crises (Pullin and Knight 2001). In this respect, Possingham (2000) argues that, 'what is intriguing intellectually is not always important from a management perspective'. Additionally, the reward system for research institutions favours short term research and rapid dissemination of results, which is likely to be incompatible with the requirements of applied conservation studies (da Fonseca 2003). On the other hand, research scientists accuse practitioners of ignoring key theoretical advances, in favour of traditional and experience based actions which have neither been systematically tried nor tested (da Fonseca 2003).

Such divisions between theoretical and practical conservation biologists have led to the duplication of conservation effort, which is not particularly helpful for the species and ecosystems under threat (Mace et al. 2000). For instance, global hotspot analyses have been carried out independently by Myers et al. (2000) and Conservation International, WWF-US

(Olson and Dinerstein 1998), the IUCN (WWF and IUCN 1994-1997), World Resources Institute (Ayensu et al. 1999), BirdLife International (Bibby 1998), and TNC (The Nature Conservancy 1997). Of concern is that each of these projects is extremely costly, and has resulted in competing priorities for conservation in addition to a variety of different methods for determining priority (Mace et al. 2000). In addition, conflicting priorities are inevitably to the advantage of those who would seek to discredit conservation actions (McShane 2003), as contradicting advice is often taken as indicative of indecision, ignorance and uncertainty and is, therefore, used as an excuse for inaction.

The generally poor degree of cooperation/coordination between researchers and practitioners results from a combination of factors, in particular: each evaluate the characteristics of a good solution differently and, consequently, put forward different priorities for conservation; intense competition for funding; biased views resulting from direct involvement in conservation activities; political persuasions and time frames; and traditional views and attitudes towards different elements of theoretical and practical conservation.

First, Levins (1966) characterised three essential properties of an 'ideal' research strategy as generality of application, degree of functional realism, and precision of expression (Fig. 10.2). However, each of these factors, while considered essential elements of a successful conservation strategy, cannot be maximised simultaneously.

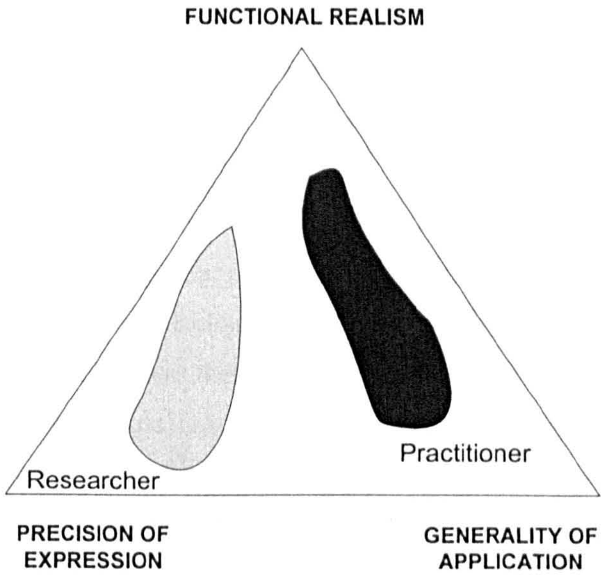

Figure 10.2: The three conflicting characteristics of a research strategy according to Levins (1966) and adapted from Baskerville (1994): precision of expression, generality of application and functional realism.

Consequently, in the initial stages of any given conservation planning exercise, decisions need to be made as regards the inevitable tradeoffs between these three factors, whilst ensuring that overall aims and objectives can be achieved in the most efficient and effective manner.

To further complicate the issue, researchers and managers weight the relative importance of each of these factors differently (Fig. 10.2). For example, wetland site managers call for general solutions, which can be readily adapted at their discretion to deal with a wider variety of conservation planning scenarios across species, sites and spatial scales, using traditional methods and experience. By contrast, researchers strive to produce clear, reasoned theories, which will stand the test of peer review.

Second, aside from the conflicting priorities and the divergent ideals of researchers and practitioners, competition for funding for conservation efforts is, more often than not, extremely fierce. However, far from being seen as vital to ensuring the quality of research, this competition is habitually counter productive (McShane 2003). Competition for funding leads to market segmentation, thereby encouraging division between theoretical and practical conservation efforts. However, the actions of those receiving support from government or public funds are likely to be tightly constrained by the fads and fashions of the day, rather than being directed by the results of scientific analysis. As a result, certain conservation activities will be more marketable at a given time than others. This, inevitably, will bias practical conservation actions towards those methods and actions that resonate best with the donor community (Slocombe 1998, da Fonseca 2003), further removing theory from practice.

Third, because investment in the development and implementation of a conservation programme may be great, practical conservation biologists are likely to be resistant to both criticism and change (Kleiman et al. 2000). Indeed, conservation organisations are often strongly opposed to evaluations of the performance of conservation actions and recommendations put forward by research institutions.

Fourth, it is rare that a conservation body has full control over a given situation as decisions are invariably influenced to a greater or lesser extent by changing political persuasions (Marren 2002). For example, the education minister who helped to set up the Nature Conservancy Council (NCC) in the UK was the same one who presided over its demise 16 years later (Margaret Thatcher). A major causative factor contributing, therefore, to the gap between research priorities and conservation actions, is that the funding available for practical conservation is dictated to a large extent by a political rather than a scientific agenda. Although, as Marren (2002) points out, election times can be especially effective for enabling conservation action. One particular example is the case of Offham Down, a chalk downland SSSI in Sussex. Specifically, following the introduction of EU grants for flax production in 1996 (£591 per acre), considerably more than for a piece of chalk downland, the site was ploughed. This would have been the end had it not coincided with a general election in 1997. Desperate to be seen as proactive, the then Secretary of State, John Gummer granted a stop order (Section 29 of the

Wildlife and Countryside Act) even though the damage had already been done. The area was then quickly restored through the actions of Sussex Wildlife Trust and FoE. Interestingly, once the order was made, the farmer himself joined in! In contrast to the positive impact of a general election on Offham Down, the fate of the internationally important Cardiff Bay estuary was effectively sealed by Labours landslide victory in 1997. Indeed, while in the opposition party, three labour MPs (Ron Davies, Peter Hain and Win Griffiths) actively protested against the barrage and pledged to stop the development if the party were elected. At the time Ron Davies said: '...they [the Conservative Government] have presided over the wholesale destruction of sites which have been designated. We have seen them wriggle dishonestly and illegally out of their international obligations under the EC directive' (Hansard 1992). In addition, Win Griffiths (Junior Welsh Office Minister) stated that: 'It is vital to retain the mudflats so that the migratory and wading birds can have the benefit of those feeding grounds' (Hansard 1992). Nevertheless, once in power and following his appointment as Welsh Secretary Ron Davies oversaw the agreements that removed the final environmental obstacles to the development. He then gave the scheme the ultimate backing when he chose Cardiff Bay as the site of the new National Assembly for Wales (http://news.bbc.co.uk/1/hi/wales/503837.stm. The European Commission is still investigating the failure of the UK Government to designate the site as part of the Severn Estuary SPA and so the legal status of the Barrage remains uncertain. In this respect the Commission has stated that in its view Cardiff Bay is an integral part of the wider Severn Estuary SPA and should therefore have been designated as such.

Finally, a general mistrust on the part of practitioners of new ideas and a reliance on traditions and experience to manage protected areas has exacerbated the lack of coordination/cooperation between conservation theorists and practitioners. Indeed, many view theoretical advancement as an unnecessary and time consuming waste of money, which could be better spent on immediate and practical conservation efforts.

Solutions

To reduce, and ultimately eliminate the gap between conservation theory and practice requires that managers are aware of and appreciate the capabilities of scientists. In their turn, scientists need to use and adapt the tools at their disposal to deliver what managers need (Prendergast et al. 1999). Ultimately, to determine the direction and purpose of applied research will require dialogue and discussion between managers and researchers prior actually to undertaking research. This dialogue should strive to reach an appropriate tradeoff amongst the three research factors (precision of expression, generality of application, functional realism) identified by Levins (1966) (Fig. 10.2; Baskerville 1994, Bunnell and Huggard 1999). Practitioners must then take the advice and suggestions presented to them as a result of this work (Possingham 2000), rather than simply dismissing them without due consideration either because the results are not particularly favourable, or the managers are unfamiliar with specific methods. In this respect, implicit in the Asia-Pacific Migratory Waterbird Conservation Strategy for 1996-2000 was the aim to resolve conflicts between working groups and local/federal government to conserve waterbirds and their habitats (Action A.4; http://www.wetlands.org).

In addition to increased dialogue, there needs to be more effort by agencies to seek help from outside their organisations. Through mutually beneficial collaborations conservation groups could increase their funding for the publication of biological data, and groups with mutual interests could collaborate on fundraising to pay for data collection (da Fonseca et al. 2000). In this respect, collaborations are beginning to be set up, for example, between CI, the Wildlife Society and WWF in east Africa. In addition, the activities of Wetlands International in general are characterised by strong interaction and collaboration with governments, intergovernmental agencies and non-government organizations in an attempt to improve cooperation and dialogue between various interest groups. For example, their activities are supported/sponsored by 140 separate organisations from 42 different countries including: the Asian Development Bank, The Philippines; the Canadian Wildlife Service; DEFRA, UK; the Disney Wildlife Conservation Fund, USA; the Global Environment Facility, USA; IUCN, Switzerland; the Living Oceans Society, USA; the Ministry for Environment and Physical Planning, Slovenia; the Parks and Wildlife Commission of the Northern Territory, Australia; the Queensland Environmental Protection Agency; the Royal Society for the Protection of Birds (RSPB); the Toyota Foundation, Japan; and the Wildfowl and Wetlands Trust (WWT), UK.

Finally, to succeed in its goals, conservation biology could take some valuable lessons from medicine. Indeed, because conservation is a crisis discipline it has often been compared to medicine in that decisions have to be made quickly, sometimes with incomplete knowledge of the situation or of the possible consequences of the action. Furthermore, like medicine conservation biology has to deal with ethical and moral issues, and conflicting goals (i.e. treatment versus side effects). Unlike conservation biology, however, in the last 25 years, there has been a re-evaluation of the ways in which medical research is carried out, in an attempt to improve overall efficiency given tight budget constraints. As a result, the focus now is on adaptive research, systematic review and evaluation of methods. A similar revolution is needed for conservation biology so that money should not be provided for carrying out biodiversity action that has little or no scientific basis (or evidence to support its effectiveness) unless the appropriate trials and monitoring are built into the action (Pullin and Knight 2001).

Mistrust of algorithms

Problems

Although there have been considerable advances in systematic approaches to reserve selection in recent years (demonstrated in Chapters 4-7), their application to real-world conservation planning remains somewhat limited. Indeed, these techniques have generally not been well received by conservation practitioners, making their application to real world conservation particularly problematic (Kingsland 2002). In consequence, most work on reserve selection remains theoretical (Prendergast et al. 1999). Priority site selection algorithms have been criticised for a variety of reasons, which fall into four general categories, namely: data, simplicity, history and tradition, and software.

First, it has been argued that reserve selection algorithms are excessively data hungry, thereby limiting their application to those areas where biological data are of exceptional quality (Davis et al. 1999, Prendergast et al. 1999, Pimm 2000, Cabeza and Moilanen 2001, Peres 2002). However, as Chapter 5 points out, the majority of datasets are inadequate/biased, even in the best-studied regions such as the UK. In particular, a common complaint is that the available data are neither good enough to use to select sites (da Fonseca 2003), nor suitable for the evaluation of existing reserves (Prendergast et al. 1999).

Theoretical conservation planning exercises within the peer reviewed literature have also been heavily criticised for the application of priority site selection algorithms to planning units (sites/areas) far larger than any available for actual conservation activities (Prendergast et al. 1999). For example, Bonn et al. (2002) used data collated for quarter degree grid cells to test whether threatened or endemic species could be used as useful surrogates for overall avian diversity in South Africa and Lesotho. This argument does not, however, apply to conservation planning exercises for waterbirds using WeBS data (i.e. Chapters 4-7) given that the site selection units relate directly to individual wetland boundaries.

Second, conservation theory is often seen as too simplistic to deal with all factors involved in a real world planning exercise (Davis et al. 1999, Prendergast et al. 1999). For example, Prendergast et al. (1999) conclude that systematic approaches to reserve selection are unable to deal with the complex and often conflicting issues of ownership, competition from other developments, and multiple demands on proposed protected areas from other political, social and economic activities. For example, individual wetlands may be in demand for farming, fishing, recreation activities (walking/boating) and bird watching. Furthermore, they argue that, because the selection and acquisition of land parcels for conservation rarely follows the structured and logical path advocated by reserve selection algorithms, often as a result of complex landscape and/or administrative structures, money would be better spend on land acquisition rather than the application of theory.

Third, there is an increasing division between those who advocate the use of systematic reserve selection approaches and those who consider expert workshops, where possible sites and conservation priorities are identified by experts from a range of disciplines, as the most effective means of identifying priorities and selecting sites for biodiversity conservation (Cowling et al. 2003b). Underpinning this polarity is the perception by many practitioners and managers that conservation is an experience-based rather than an evidence-based activity (Pressey 1999, Pullin and Knight 2001, Cowling et al. 2003a, 2003b). Indeed, efficient solutions (see Chapters 4-7) may not be sought if managers believe that this can be achieved in other ways (Prendergast et al. 1999). For example, one recognised alternative to awarding protected status to priority areas for conservation is the translocation of threatened species to alternative locations, as was the case for Desmoulin's whorled snail (*Vertigo moulinsiana*), whose presence threatened to impede road developments in England (see Marren 2002). The argument is that priority species can simply be moved from threatened areas, either to those already covered by formal protection agreements or those available for conservation. This is despite the recognition that the majority of such translocations inevitably fail (Marren 2002). This translocation approach was also attempted to a certain extent following the flooding of the mudflats at Cardiff Bay. As a concession for the loss of this valuable estuarine habitat, 1,000 acres of farmland were flooded with sewage water 15 miles further along the coast (the Gwent Levels Reserve). From a conservation perspective, however, the creation of freshwater marshland does not provide suitable alternative habitat for estuarine species. Indeed, although the aim was to provide alternative feeding/roosting areas for displaced species, there have as yet been no records of ringed birds from Cardiff Bay appearing on the newly created Gwent Levels reserve (http://www.foe.co.uk/resource/press_releases/19970707161413.html).

Fourth, many practitioners have rejected systematic approaches for their complexity, cost, and inaccessibility. Indeed, the cost of site selection software, and in particular, linear programming optimisation software, is often used as a reason (excuse) for the use of traditional approaches.

Solutions

All datasets are to some greater or lesser extent imperfect (Pressey and Cowling 2001, Niesenbaum and Lewis 2003; Chapter 1 for a discussion of the limitations of WeBS) and although additional data collection is highly desirable, this will likely require more time than is currently available, especially for threatened species (Chapter 5; Niesenbaum and Lewis 2003). However, perfect information is not a pre-requisite for the use of complementarity-based site selection algorithms (Pressey and Cowling 2001, Ferrier 2002, Rodrigues 2002). What is required, however, is that data for all candidate sites are of equal quality and composition, and that the relevant information is spatially referenced (Margules and Pressey 2000, Williams et al. 2002), as is the case for the WeBS abundance records (Chapter 1). Inevitably, there is no algorithm capable of producing good quality results from exceptionally bad data (the Garbage In Garbage Out rule of computational science; Rosing et al. 2002). It is, therefore, important that any potential sources of error or bias within data sets be appropriately accounted for in the application of these mathematical approaches wherever possible to make the best use of existing data, rather than concealing them within a subjective planning exercise (Rodrigues 2002). Specifically, throughout this thesis, only those sites with ≥60% of the abundance records present were included within subsequent analyses, and any remaining missing counts were imputed using linear interpolation (see Chapter 1).

To provide the opportunity for successful conservation actions requires that both expert knowledge and algorithms are included throughout a systematic conservation planning exercise. Taken alone, neither approach is sufficient. In particular, expert judgement is essential for setting conservation priorities and targets for representation (Chapters 4-7), raising awareness of key issues (e.g. potential management conflicts; Chapter 7), discussing the requirements of various interest groups and stakeholders (e.g. fishermen, farmers and bird watchers), and facilitating compromise and consensus (McCoy et al. 1999, Asquith 2001, Cowling et al. 2003b). The major disadvantages of such an approach are: bias relating to the particular interests, knowledge and preferences of experts involved; and subjective, non-repeatable or transparent, time-consuming decisions, following the path of least resistance (choosing sites with the highest probability of being purchased) as opposed to conservation value and complementarity. Indeed, priority sites identified using a panel of experts have been shown to be considerably less effective at representing biodiversity than those sites selected using systematic site selection algorithms (e.g. Chown et al. (2001), using data for biodiversity in the southern oceans, and Cowling et al. (2003b), using data for threatened flora for the Cape floristic region of South Africa). Furthermore, because Cowling et al. (2003b) employed a greedy heuristic approach (Chapter 4) for the selection of priority conservation areas, it is likely that the performance of the expert selection was even worse.

In contrast to expert driven approaches, systematic reserve selection methods have the advantage that they are transparent, repeatable, accountable, flexible, target-driven and objective (as shown in Chapters 4-7). Furthermore, given the uncertainty associated with the conservation of biodiversity, the ability repeatedly to test hypotheses by computer modelling is essential, especially in view of the likely implications of a poorly designed reserve network on the persistence of biodiversity (Kingsland 2002). In the context of wintering waterbird conservation in Great Britain, these approaches are invaluable for the evaluation of the performance of the current protected area network (Chapters 2 and 3), the exploration of alternative priority site selection criteria (Chapter 4), the incorporation of species specific representation targets (Chapters 4-7), the identification of gaps in coverage (addressed in Chapter 4), the exploration of alternative network configurations (Chapter 6) and the incorporation of possible management conflicts between individual priority species (Chapter 7). More generally, the site selection methods tested and refined through the assessment of the SPA/Ramsar Site network in Great Britain can be adapted and applied to other conservation planning scenarios elsewhere in the world (discussed in detail in Chapter 5). Furthermore, in recent years, systematic approaches to reserve selection have been modified to account for various real-world constraints, including species persistence (Cabeza 2003), opportunity costs and other economic factors (Faith et al. 2003), and protected area connectivity (Briers 2002), thereby adding to their utility for conservation planning. Such advances are particularly advantageous (should they be adopted that is!) for multinational conservation programmes such as the African-Eurasian Waterbird Agreement (AEWA) or the North American Waterbird Management Plan (NAWMP), which aim to make provisions for migratory waterbird species across distinct political and geographical boundaries. Indeed, provided suitable data are available, linear programming algorithms could be instrumental in the identification of interconnected networks of sites to take account of the diverse requirements of individual species (i.e. breeding grounds, migratory pathways, stop-over sites and wintering areas) in the most effective and efficient manner. In this way, planners can be provided with a clear, flexible set of possible protected areas as a catalyst for negotiations over land acquisition, protected area designation, stakeholder requirements/wishes, and site/network management strategies (discussed in Chapter 6).

A common criticism of sophisticated reserve selection software for practical conservation planning is the cost of software acquisition. Indeed, it is for this reason that, where selection algorithms have been employed, cheaper iterative heuristic approaches have often been favoured over linear programming techniques. However, given the likelihood of achieving sub-optimal solutions to reserve selection problems through the use of iterative heuristic methods (Rodrigues and Gaston 2002, Önal 2004), wherever possible (i.e. where a problem can be expressed as a linear, integer formulation), optimisation algorithms should always be the tools of choice. In recent years, there has been an expansion of software (both heuristic and optimisation) available over the Internet, for example, CPLEX™ (ILOG 1999), WORLDMAP™ (Natural History Museum 1998), CODA™ (Bedward et al. 1992) and MARXAN™ (Ball and Possingham 2000). Although these trial versions lack many of the full versions' advanced capabilities they are operational and ideal to test their applicability for research projects and management plans.

The lower costs of sub-optimal conservation planning software or a reliance on traditional methods as compared with the more expensive linear programming optimisation software, will ultimately only be a financial benefit in the short term. Indeed, given that the former methods are unlikely to identify all the most suitable sites for biodiversity conservation (i.e. an optimal reserve network; Chapter 4), adding additional sites to the network in the future will result in higher overall expenditure compared with efficient networks selected using optimisation software. For example, Chapter 4 identifies a set of ten sites that would have improved the performance of the 1998/99 SPA/Ramsar Site network by a greater amount than that realised through the 82 wetlands actually added (new sites classified/designated after 1998/99). Thus, the costs of land acquisition and long term management of a network comprising 220 sites will certainly be greater over time than that of an efficient network selected using the initially more expensive optimisation software.

Knowledge

Problems

Public opinion is a powerful motivating force for government action and is consequently a factor in funding allocation for activities. As a result, one of the most pressing problems for the practical application of conservation theory is a lack of public knowledge and awareness of key issues. For example, only 26% of those questioned by DEFRA in the 2001 'public attitudes to the environment' survey (DEFRA 2001) recognised the term 'biodiversity'. Moreover, environmental issues were low on the list of factors thought to affect quality of life after, money, health, crime, job, neighbours/neighbourhood, transport and housing. In consequence, because some areas of conservation are more popular than others (mainly through a lack of awareness of other issues), and because conservation agencies and NGOs are reliant on public support, conservation action may not always reflect priorities identified by research institutions.

A second problem facing conservation practitioners is that the majority of theoretical conservation papers are published in English (Primack 2001). This practice denies invaluable research to many practitioners, and importantly to many students who do not speak or read English. The problem becomes even worse as the majority of areas where biodiversity is seriously threatened are in those areas where English is not widely spoken and where the finances available to provide instruction in English are not available.

In addition to this language barrier, in many less developed countries, the availability of key journals such as *Animal Conservation*, *Biodiversity and Conservation*, *Biological Conservation*, *Conservation Biology*, *Ecology* and *Journal of Animal Ecology* is limited due to lack of funds and inadequate technology (Finch and Patton-Mallory 1992, Prendergast et al. 1999). Furthermore, many of the findings relating to the success or failure of active management are confined to internal reports for various organisations, and as a result are difficult to find.

Solutions

Raising awareness of the most pressing conservation problems will certainly go a long way towards reducing the disparity between conservation priorities identified by researchers, and the activities of practitioners receiving funding (James 2002). Not only would this have to be carried out for the general public, but also at the undergraduate level through interdisciplinary teaching. In this way, new generations of conservation biologists will be better able to communicate their ideas, both to the public and also to conservation practitioners or researchers. In this respect, WWF-UK challenged the British government over the lack of public consultation/involvement in the Cardiff Bay barrage review in the European court of First Instance on the basis of the Code of Conduct Concerning Public Access to Commission and Council Documents (93/730/EC). The British government, however, gave evidence in support of the Commission's case that these documents should remain confidential, and claimed that release of such information would prejudice the achievement of negotiated settlement of disputes (http://www.wwf.org.uk). There are, nonetheless, several ways through which improved information dissemination can be achieved.

First, the Internet not only provides an excellent opportunity for raising funds and awareness, but also to link countless, widely dispersed conservation practitioners with each other and with the academic community, conservation organisations, government land agencies, and local communities (Adams et al. 2002). In this respect, many of the key conservation journals are now available online. In addition, many national and international conservation organisations provide a wealth of clearly presented, detailed and up-to-date information. Inevitably, if the Internet is to provide such a service, information must be accurate, regularly updated, and report failures as well as successes (Redford and Taber 2000). For example, Wetlands International's website provides details of their ongoing activities, as well as facts and figures relating to threatened species, relevant conservation conventions (Ramsar Convention, Bern Convention, Bonn Convention) and key Directives (Birds Directive, Habitats Directive), conservation activities by region and key issues of general interest (http://www.wetlands.org). Similarly, websites provided by the British Trust for Ornithology (BTO) (http://www.bto.org), RSPB (http://rspb.org.uk) and WWT (http://www.wwt.org.uk) provide information relating to specific research activities, membership details, general conservation issues, local events and species of national conservation concern. Nonetheless, many field practitioners remain resistant to the powers of the Internet, as it is perceived that there are always crises more deserving of scant conservation funds than the purchase of expensive computer hardware and software (Adams et al. 2002).

Second, corporate sponsorship will not only be of benefit to the species in question, but also the company providing the funds. Particular examples of corporate sponsored Species Action Plans (SAPs) include: depressed river mussel (*Pseudanodonta complanata*) sponsored by The Environment Agency; bullfinch (*Pyrrhula pyrrhula*), deptford pink (*Dianthus armeria*), shore dock (*Rumex rupestris*), convergent stonewort (*Chara curta*), lesser-bearded stonewort (*C. curta*), slender stonewort (*Nitella gracilis*), starry stonewort (*Nitellopsis obtusa*), tassel stonewort (*Tolypella intricata*) and great tassel stonewort (*T. prolifera*), and the churchyard lichen (*Lecanactis hemisphaerica*) sponsored by Centre Parks; medicinal leech (*Hirudo medicinalis*) sponsored by Glaxo Wellcome; bittern (*Botaurus stellaris*) sponsored by the cooperative bank; large blue butterfly (*Maculinea arion*), pearl bordered fritillary (*Boloria euphrosyne*) sponsored by ICI; roseate tern (*Sterna dougallii*) sponsored by Northumberland Water; corncrake (*Crex crex*) sponsored by Shanks McEwen; skylark (*Alauda arvensis*) sponsored by Tesco; and otter (*Lutra lutra*), early gentian (*Gentianella anglica*), speckled footman moth (*Coscinia cribraria bivittata*), heath fritillary butterfly (*Melitaea athalia*) sponsored by Fina/Wessex water (Marren 2002). Despite the variety of species currently supported, the lack of sponsorship for individual waterbird species (with the exception of roseate tern) should be noted, particularly given their international conservation importance.

Third, a simple solution to the language barrier would be for English speaking scientists to seek out scientists from other countries to translate their publications into other languages (Primack 2001). For example, all abstracts for articles published in *Conservation Biology* are presented in Spanish as well as in English, although the accuracy of these translations is often questioned. In addition, foreign scientists could add relevant examples into texts to facilitate understanding. For example, texts concerned with migratory waterbird conservation published in the UK ideally contain details of individual SPAs or Ramsar Sites and the measures taken regarding their selection and management. Similarly, books intended for distribution in the USA should incorporate information as regards the North American Waterbird Management Plan (NAWMP). This translation may not necessarily involve a monetary cost, as institutions will be willing to supervise translations to get the textbooks (Primack 2001).

Finally, as a means of encouraging cooperation between various organisations and practitioners, as well as improving information dissemination, there are various international conservation-oriented conferences each year, many of which have grants and awards for overseas participants and students (e.g. SCCS, British Ecological Society (BES) and SCB), devoted to different aspects of conservation and management, which are attended by many members of NGOs and government conservation bodies.

Circular arguments/Lack of consensus

Problems

Theoretical conservation biology has been dogged since its inception by lack of consensus over definitions for basic biological terms (i.e. biodiversity or naturalness), which make it extremely difficult to translate theory into practice (Sarkar and Margules 2002, Heywood and Iriondo 2003). Additionally, conservation biology has been characterised by persistent circular arguments concerning key principles (i.e. the Single Large Or Several Small (SLOSS) debate over protected area configuration), which complicate the implementation of theory on the ground. In recent years, consensus has been reached that there is no universal decision rule to determine the configuration of individual protected areas (Kingsland 2002). More recently, however, a new debate concerning the use of expert opinion versus priority site selection algorithms has emerged, which is threatening to divide conservationists yet again (discussed in detail in the preceding section).

Solutions

Increased cooperation and dialogue between researchers and managers will facilitate consensus regarding definitions and key principles. Nonetheless, practitioners must tell the researchers what needs to be investigated and the practitioners in their turn must be able to take the advice and suggestions presented to them as a result of this work (Pullin and Knight 2001). Workshops involving conservation theoreticians and practitioners from different countries will be instrumental in providing consensus over definitions and principles, providing of course that they themselves can agree!

Stakeholder involvement/social agendas

Problems

Attitudes towards conservation vary greatly depending on how the individual is affected by particular actions and decisions (Marren 2002). Nonetheless, research has shown that a notable proportion of the public regularly argue in favour of environmental issues and express interest in conservation initiatives (James 2002). Their actions, however, often indicate otherwise. For example, many have argued for sustainable energy sources such as windmills, as an alternative to the continued use of fossil fuels as long as there is no disruption to their immediate surroundings and quality of life (NIMBYism, the 'not in my back yard' attitude). Indeed, like almost everyone else, private landowners value financial security and personal freedom over conservation (da Fonseca 2003). This is clearly demonstrated by the conflicts and concern shown over the designation and regulation of SSSIs in the UK (see Chapter 1). Thus, NIMBYism is likely to affect the success of proposed conservation activities, particular if they are likely to affect the daily lives of local communities (e.g. the flooding of farmland for the creation of the Gwent Levels Reserve; set aside land to comply with SSSI designations; cessation of fishing rights; restricted access to sensitive areas).

Often, however, the reason for the failure of conservation activities is that local people and stakeholders (i.e. private landowners) have not been properly informed, involved or consulted during the planning phase. For example, returning again to the flooding of Cardiff Bay, the Welsh Secretary Ron Davies conducted a confidential review into the proposed barrage without participation from the public, local farmers or conservation organisations. This course of action particularly angered local farmers as land for the new Gwent Levels reserve was obtained under compulsory purchase. In addition, given the area was flooded using sewage water the fear was that polluted pools would affect groundwater quality and damage surrounding farmland. Inevitably, various groups/organisations will have a vested interest in the ways in which individual wetlands areas in Great Britain are managed and which sites should be set aside for formal protection, namely: private landowners, farmers, gamekeepers, wildfowlers, fishermen, hikers, climbers, water sports enthusiast and bird watchers. In consequence, given the conflicting demands on individual sites from each of these interest groups, dialogue in the early stages of a planning process is essential to reach a satisfactory compromise. For example, rock faces are considered vitally important for various nesting sea birds such as fulmars (*Fulmarus glacialis*) and kittewakes (*Rissa tridactyla*), but are also in demand by rock climbers. Alternatively, large standing waterbodies are invaluable for many wintering waterbird species (i.e. mallard, pochard, goldeneye and coot), however, they are also prime locations for jet skiing, windsurfing and sailing. Without doubt, these recreational activities will disturb and distress feeding and/or roosting birds in the vicinity, nonetheless, they are legitimate and popular pursuits and must, as a result, be satisfactorily accounted for in the planning phase.

Essentially, parks that have been designated in areas where people live and who have not been included in the planning process are unlikely to be supported by locals. For example, the creation of Madagascar's Ranomafana national park, which encompasses areas used by local people for subsistence, was carried out without either consulting or informing these people. As a direct result of this failure, the area now experiences social conflict, economic distress and a distrust of foreign scientists and conservation schemes (James 2002). In contrast, taking heed of such warnings and in a bid to engage local communities in Indonesia, some conservation projects have been hijacked by a social agenda. As a result, large sums of money, ostensibly assigned to conservation, have never reaped measurable conservation benefit (Whitten et al. 2001).

Solutions

In the past, conservation activities have been uncoordinated with the interests of private landowners and local people (Tikka and Kauppi 2003). If conservation actions are to succeed, these two groups must be involved in the planning process from the start. Inevitably, the management of protected areas will always be an easier task when local people are actively involved (Kingsland 2002). Nonetheless, it is essential that local communities given a stewardship role over protected areas must be prepared and able to take on the task (Kingsland 2002). In addition, top-level commitment (uncoupled from the promises made during national/local elections) is essential to ensure that all involved take conservation proposals and designations seriously. Without senior commitment it will be much harder to give the planning process and those working within it the power to make any difference (Slocombe 1998). In this respect, workshops and open meetings need to be an integral part of any planning process (Cowling et al. 2003b). The result of this involvement is the better use and permanence of protected areas that fully comply with the objectives set out during the planning phase (Arias and Valery 1999). There have been some success stories, in particular, the establishment of a new national park on Northern Bathurst Island as a result of a partnership between the Canadian Nature Federation (CNF) and the Mining Association of Canada (MAC) (Spence and Gratton 2003). Jointly, the two organizations successfully proposed a solution to the difficult question of how to protect Peary caribou habitat at Northern Bathurst Island while keeping opportunities open for the mining industry in the area.

10.3 The role of conservation biology

Aside from an uncertain future, a reality of conservation biology is that it must take account of many factors other than biodiversity (Faith and Walker 2002). Indeed, effective conservation planning must combine a variety of techniques, including expert knowledge, policy development, legislation, taxonomy, GIS, statistical modelling, negotiation, compromise and experience. It will, out of necessity require the integration of information from many different disciplines. The skills required to succeed include the ability to think critically, communicate orally and in writing, participate in group decision making, advocate for conservation, translate science to the lay-public, and work within political, social, economic and cultural contexts (Niesenbaum and Lewis 2003).

The analyses presented throughout this thesis represent one step towards achieving fully informed, unbiased conservation planning, making the best use of the data available at a scale that can be applied to actual conservation efforts on the ground. However, what is now necessary is for the conclusions and recommendations presented here to be applied not only to the existing network of SPAs and Ramsar Sites in the UK, but also, where appropriate, for the selection of protected areas particularly for waterbirds elsewhere in the world. In an attempt to solve or at the very least limit the various problems outlined above (Section 10.2), conservation biology and conservation biologists will have to take on the diverse, difficult and often controversial roles of unbiased and focussed mediator, collaborator, educator and publicist.

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APPENDIX I

Dealing with population fluctuations in waterbird conservation

S.F. Jackson[1], K.J. Gaston[1] and M.Kershaw. [2]

[1]Biodiversity and Macroecology Group, Department of Animal and Plant Sciences,
University of Sheffield, Sheffield S10 2TN, U.K. E-mail: s.jackson@sheffield.ac.uk
[2]The Wildfowl and Wetlands Trust, Slimbridge, Gloucester GL2 7BT, U.K.

Abstract

The selection of sites as Special Protection Areas (SPAs) in Great Britain is a specific requirement of the European Union (EU) Birds Directive. This states that waterbird populations/assemblages that are listed as 'interest features' on these SPAs should be maintained or restored to a 'favourable conservation status'. At a site level, conservation objectives are set for each interest feature and these include targets for population size such that numbers should not fall significantly below the current level. The expectation, therefore, is that these protected sites will, on average, hold greater numbers of individuals for each target species compared to non-designated sites and, because of the imposition of a lower bound on population sizes, a lower variance in numbers over time. To test these assumptions we compared the populations of 21 selected waterbird species on SPA and non-SPA wetlands in Great Britain in relation to their mean population size and associated variance over a six-year period (1993/94-1998/99). SPAs generally have larger population sizes compared with the non-SPAs. However, there was no significant difference in population variance, most likely due to the influence of large-scale processes on local population dynamics. In consequence, whereas management of sites in isolation may influence the total numbers of birds wintering at a particular site, such an approach is unlikely to be effective in the regulation of population variance in numbers.

Introduction

Wetlands in Great Britain attract millions of migratory waterbirds each winter, many of which are of arctic and north temperate nesting species, making their conservation a matter of both national and international concern. In this respect, wetlands and their associated avian populations are specifically targeted by the European Union (EU) Birds Directive (Council Directive 79/409/EEC), which stipulates that European Union Member States take active measures for the conservation of all species of wild birds found naturally within their boundaries. Additionally, particular attention is required for those species considered to be especially vulnerable (listed in Article 4 of the Directive). The primary measure adopted to achieve these objectives is the classification of important sites as Special Protection Areas (SPAs), many of which are wetlands. In the UK, selection of SPAs has two stages, and is intended to be implemented on a site-by-site and species-by-species basis following specific guidelines published by the Joint Nature Conservation Committee (JNCC) in 1999. In essence, Stage one identifies suitable sites for SPA classification based on numerical thresholds for population size (JNCC 1999). Following the application of Stage one guidelines, Stage two requires that the specific ecological factors of the proposed sites be evaluated, to further refine the selection, including breeding success and naturalness. Accordingly, those sites considered most suitable should eventually be classified as SPAs.

In recognition of the not insubstantial time and resource requirements of the SPA selection process, it is of paramount importance that the most suitable wetland sites, in terms of the requirements of target species, are identified. In addition, because of the considerable difficulties associated with revoking protected area status, it is crucial that selected sites are those best equipped to sustain viable populations into the future. Not only is this vital in light of increasing pressures on existing wetlands for anthropogenic advancement, which are likely to diminish the opportunities for conservation in the future, but also so that planners and managers can confidently justify a site's inclusion in the network should conflicts concerning constituent sites arise. However, how to actually ensure species persistence remains one of the central themes in conservation planning.

With regards to the EU Birds Directive, an explicit requirement of signatories is to ensure populations of target species (as defined by the Directive, Article 4) do not decline once a site is awarded SPA status (Article 13). Furthermore, this should be achieved primarily through the implementation of site-specific management schemes. In effect, this means that for each SPA, there is a lower limit on the population sizes of all target species. As a consequence, to successfully maintain the populations of individual species above these baseline levels site managers can only realistically take measures to either increase the mean population size, or to decrease the variance in numbers. However, these two management options are not mutually exclusive, indeed, by attempting to maintain mean population numbers above initial levels, it is likely that there will also be a reduction in the variance over time.

In light of the imposition of a lower bound on population size under legislation at EU level and as a consequence of site management, the expectation is that waterbirds in the UK will exhibit greater mean population sizes and lower magnitude population fluctuations over time when compared with non-protected wetlands. To test these assumptions, we compared populations of 21 waterbird species on SPA (classified prior to 1990/91) and non-SPA wetlands in Great Britain in relation to their mean population size and the associated variance over a six-year period (1993/94-1998/99).

Methods

Data

Analyses were conducted for 21 species of waterbird using data from the Wetland Bird Survey (WeBS). This scheme, a joint venture by the British Trust for Ornithology (BTO), the Wildfowl and Wetlands Trust (WWT), Royal Society for the Protection of Birds (RSPB) and the Joint Nature Conservation Committee (JNCC), aims to monitor the populations of non-breeding waterbirds in the UK (Kirby 1995, Cranswick *et al.* 1997, Pollitt *et al.* 2000). In brief, it relies on bird counts provided by a network of volunteers from a range of wetland types visited on a co-ordinated date each month, with emphasis on winter months September to March. More than 2000 wetlands are included in the scheme annually, and in excess of 8000 have been counted at least once since 1960 (Cranswick *et al.* 1997).

We included data for the years 1993/94 to 1998/99 (years for which there were sufficient numbers of sites with complete counts) inclusive, for Little Grebe *Tachybaptus ruficollis*, Great Crested Grebe *Podiceps cristatus*, Cormorant *Phalacrocorax carbo*, Whooper Swan *Cygnus cygnus*, Bewick's Swan *Cygnus bewickii*, European White-fronted Goose *Anser albifrons albifrons*, Dark-bellied Brent Goose *Branta bernicla bernicla*, Shelduck *Tadorna tadorna*, Mallard *Anas platyrhynchos*, Gadwall *Anas strepera*, Pintail *Anas acuta*, Shoveler *Anas clypeata*, Wigeon *Anas penelope*, Teal *Anas crecca*, Pochard *Aythya ferina*, Tufted Duck *Aythya fuligula*, Goldeneye *Bucephala clangula*, Smew *Mergellus albellus*, Goosander *Mergus merganser*, Red-breasted Merganser *Mergus serrator* and Coot *Fulica atra*. These 21 waterfowl species are those considered well represented by WeBS.

To allow for accurate comparison, the SPA network was taken as the 79 sites classified as SPAs prior to and including 1990/91. In this way, there were sufficient numbers of sites included in the analysis, the numbers of SPAs remained constant for all years, and sufficient time has elapsed for management actions to take effect. Although there are data available for Northern Ireland these analyses use data from wetlands in Great Britain only. The numbers of non-SPAs used in this analysis varied between each species and ranged from 88 to 1724.

Analyses

To assess the performance of the existing wetland SPA network compared with non-SPAs, peak annual population counts for each of the 21 species were calculated for each site individually. For each year (from 1993/94-1998/99), these site counts were taken as the peak annual population count at that site. For each species, the mean population size and variance in numbers were calculated for each individual site over the six-year period. These values were then log transformed to account for the wide range in the mean and variance for individual sites. To examine the effects of temporal trends in the population numbers between 1993/94 and 1998/99, all analyses were conducted both using raw and detrended variance data (detrended using a quadratic trend model (Minitab™ Version11.12)). However, because the results were largely identical only the results of the analyses using detrended data are reported here. To account for the mean dependence of population variance in numbers, mean-variance comparisons were made using Analysis of Covariance (ANCOVA), mean population size as the covariate, population variance as the dependent variable, and site type as the fixed factor. We recognise that not all of the species present on a protected site will be the reason that the site was classified (interest features), which, because managers are not legally required to provide measures for the protection of such populations, is likely to confound the results of these analyses. However, because a large proportion of SPAs were classified for their overall waterbird assemblages in addition to individual species these analyses were completed using all SPAs, irrespective of whether a species was a listed feature on the site or not.

Results

Contrasting SPA and non-SPA population sizes

Individually, the distribution of the SPAs, in terms of their mean population size, varied considerably between species. However, contrary to expectation given the numerical threshold methods used to select sites, for 18 of the 21 species less than 50% of the SPAs were ranked within the top fifty sites for population size (sites ranked separately for each of the 21 species). In contrast, for the remaining three species (Bewick's Swan, European White-fronted Goose and Dark-bellied Brent Goose) greater than 50% of the SPAs were in the top fifty. Nonetheless, despite the low percentages of SPAs ranked in the top 50 sites for population size, for 19 of the 21 species the frequency distributions of the mean population sizes were significantly different (two-sample Kolmogorov-Smirnov test) between SPAs and non-SPAs. Indeed, in general, for SPAs the distribution of mean population sizes was significantly right skewed and sites tended towards the larger population sizes. By contrast, for non-SPAs the majority of sites supported small populations only (Figure 1a). Conversely, for Whooper Swan and Goosander, there were no significant differences between the frequency distributions of the SPA and non-SPA wetlands (Figure 1b).

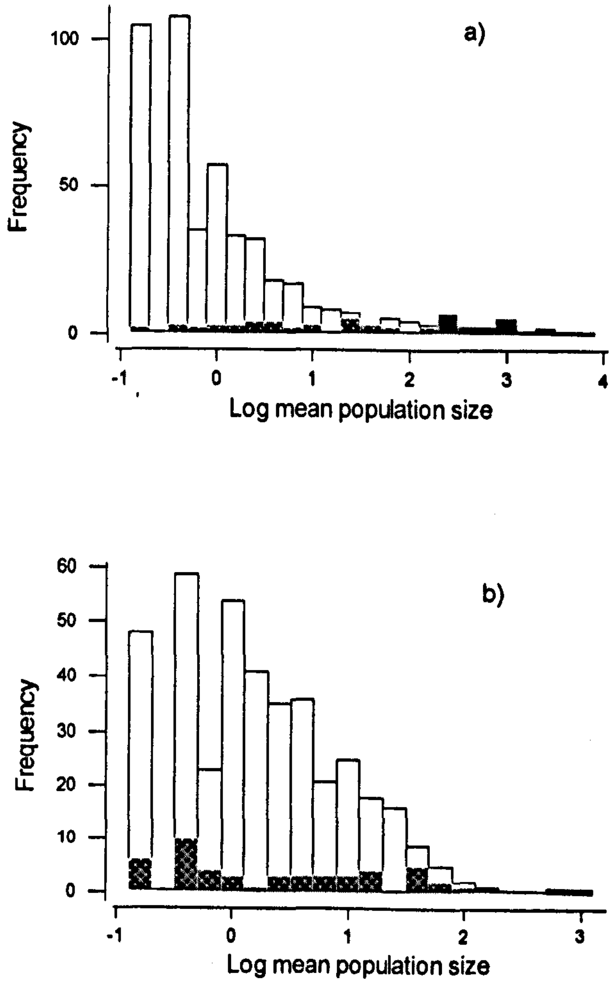

Figure 1. Frequency distributions showing the mean population sizes, calculated between 1993/94-1998/99, for sites classified as SPAs (black, filled bars) and non-SPAs (empty bars) for Pintail (a) and Whooper Swan (b). The overlap between the two site types is shown by the dark cross hatched bars.

As intuitively expected, the percentage of SPAs in the top fifty sites for population size correlated positively with the aggregative tendency (defined as the percentage of the national population which is found in the top ten sites for population size) of a species (r = 0.536, d.f. = 21, $P < 0.05$) (S. F. Jackson, unpublished analysis). Specifically, species which tend to form dense flocks, such as for example European white-fronted goose and Bewick's swan, generally have a greater percentage of the SPAs within the top fifty sites. In contrast, species with a more dispersed distribution pattern, such as tufted duck and goldeneye, tended to have a significantly lower percentage of their SPAs ranked within the top fifty sites for population size.

Population mean-variance comparisons for SPAs and non-SPAs

As expected, there was a strongly significant positive correlation between mean and variance in population numbers under logarithmic transformation for both the SPAs and the non-SPAs for each of the 21 species (for examples see Figure 2). Indeed, r^2 values ranged from 0.68 to 0.98 for SPA wetlands, and from 0.76 to 0.95 for non-SPAs.

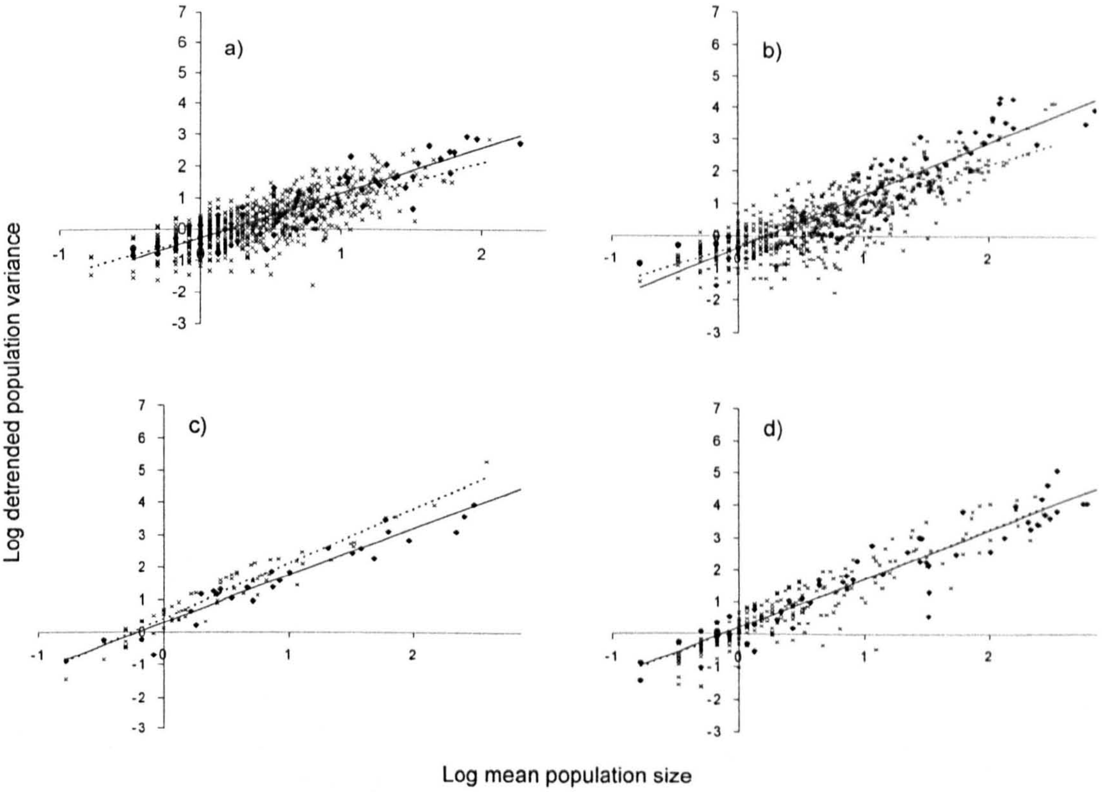

Figure 2. The relationships between the log detrended population variance and the log mean population size for sites classified as SPAs (diamonds) and for non-SPAs (crosses) calculated between 1993/94 and 1998/99 for each individual species: a) Little Grebe, b) Great Crested Grebe, c) Bewick's Swan and d) Pintail. The lines represent the linear trends for SPAs (solid line) and non-SPAs (dashed line).

Considering only the 18 species for which there were no significant differences in the slope of the mean-variance relationship (an explicit requirement of the ANCOVA procedure) for SPA and non-SPA populations (Table 1), intercept comparisons (population variance) for 15 of the 18 species were not significantly different ($P >$ 0.05). Only Dark-bellied Brent Goose, Wigeon and Tufted Duck showed significant differences, although for each this was borderline (Table 1). Moreover, for each of these three species, the population variance was greater for SPA wetlands compared with non-SPAs. In conjunction with this lack of significant difference between SPA and non-SPAs, there was no significant difference in the numbers of times the SPA slope was above that of the non-SPAs ($P > 0.05$), tested using the paired t-test.

For Great Crested Grebe, Bewick's Swan and European White-Fronted Goose there were significant differences ($P < 0.001$) between the two site types in terms of the slope of the mean-variance relationship, which excluded these species from the ANCOVA analysis. Nevertheless, for Great Crested Grebe and Bewick's Swan the population variance for a given mean population size was generally, greater for SPAs compared with the non-SPAs. The reverse was true, however, for European White-fronted Goose.

Table 1. The ANCOVA summary table showing the F test for equality of slopes for each species and the F values for intercept comparisons for each of the 21 species. The ANCOVA was carried out using log detrended population data, with mean population size as the covariate, population variance as the dependent variable and site type (either SPA or non-SPA) as the fixed factor. * $P < 0.05$, ** $P < 0.01$.

Species	Equality of slopes	Intercept comparisons
Little Grebe *Tachybaptus ruficollis*	4.470	3.902
Great Crested Grebe *Podiceps cristatus*	20.607**	
Cormorant *Phalacrocorax carbo*	0.133	2.054
Whooper Swan *Cygnus cygnus*	0.024	1.324
Bewick's Swan *Cygnus bewickii*	13.210**	
European White-fronted Goose *Anser albifrons albifrons*	21.559**	
Dark-bellied Brent Goose *Branta bernicla bernicla*	2.271	5.519*
Shelduck *Tadorna tadorna*	9.197	1.863
Mallard *Anas platyrhynchos*	1.962	0.089
Gadwall *Anas strepera*	6.038	0.559
Pintail *Anas acuta*	0.343	0.114
Shoveler *Anas clypeata*	0.112	0.639
Wigeon *Anas penelope*	0.001	4.663*
Teal *Anas crecca*	0.020	6.989
Pochard *Aythya farina*	2.996	0.407
Tufted Duck *Aythya fuligula*	1.038	4.488*
Goldeneye *Bucephala clangula*	7.302	0.469
Smew *Mergellus albellus*	0.049	0.138
Goosander *Mergus merganser*	0.282	0.215
Red-breasted Merganser *Mergus serrator*	0.345	0.398
Coot *Fulica atra*	2.908	2.388

Discussion

For SPAs in Great Britain, although it seems intuitively sensible to advocate conservation measures to maintain target populations at levels at or above those prior to classification, the practical implications of such a suggestion are complex, particularly in light of the current isolated-sites approach to management. In this respect, for the species analysed here, such a management strategy is likely to be confounded by the influence of large-scale processes on the local population distributions. Indeed, the distribution of a species on individual sites is determined not only by local factors such as site quality, but also by large-scale demographic factors, such as weather conditions and resource availability at breeding sites. Despite this, however, the mean population sizes for 19 of the 21 species are generally greater on the SPAs compared with the non-SPA wetlands (see Figure 1a). Although this is perhaps unsurprising given the numerical population thresholds used to select these sites (for details see JNCC 1999), it is, nonetheless, highly encouraging. Alternatively, the generally larger mean population sizes observed for SPAs may be, in part, due to the efforts of the individual site managers and not simply a result of the SPA selection procedure. It is not unreasonable, therefore, to assume that site-based management regimes can, to some extent at least, influence the total numbers of birds wintering at any particular site. For example, by increasing the size of a particular lake, a greater proportion of the wintering population in Great Britain could over winter on a particular wetland.

By contrast, for the 21 species analysed here it would appear that the isolated sites approach to SPA management has failed to make any discernible difference to the variance in numbers of the individual species. However, aside from the fact that it may simply be that site managers may not in fact be targeting waterbird population fluctuations as part of a management strategy, there are several reasons why this may be the case. Firstly, it may be that the measures undertaken to protect and maintain waterbirds are not in fact having the desired effect, either because not enough is currently known about what is actually required, or that the intensity of such activities is insufficient to produce a discernible effect. Alternatively, for sites with more than one official designation, or where more than one species is listed as a management priority under the SPA selection guidelines, conflicts of interest are inevitable. Indeed, it may be that actions taken to favour one species are directly in opposition with the requirements of another, such as provisions for diving versus dabbling species. In addition, actions aimed at the management of whole assemblages may also have a negative impact on the populations of individual target species.

It is, however, more likely that difficulties with the actual practical application of the Birds Directive are directly responsible for the lack of significant difference between SPA and non-SPA wetlands, rather than inadequacies in site management. Indeed, with regards to the variance in numbers over time, for migratory species the dynamics of individual species are likely to be the result of many processes operating at different spatial scales (Thomas & Kunin 1999, Gaston & Blackburn 2000). In this respect, the overall lack of significant difference in population variance between non-SPAs and SPAs for 18 of the 21 species (Table 1) most likely results in major part from the characteristics of migratory species themselves. Because waterbird populations are not static and the birds are capable of moving great distances, in many cases greater than 1000km, management is not easily confined to isolated sites (Storrs & Finlayson 1997, Esler, 2000, Turner *et al.* 2000). This dominating effect of large-scale processes is also reflected in the lack of any significant correlation between January counts in different years between all possible pairs of sites (S. F. Jackson, unpublished analysis). In consequence, because of the disparity between the scale of demographic processes and that of management regimes, consideration of sites in isolation is unlikely to be especially effective for the control of annual population fluctuations.

As a final possibility, although it is recognised that some errors in these data inevitably exist, it is unlikely that these alone will be responsible for the similarity, in terms of the variance in numbers, between SPAs and non-SPAs. The data currently available through WeBS are generally accepted as the most accurate and comprehensive available, and unlike the greater majority of biological census data, are systematically collected by volunteers at set times and dates throughout the winter season. In addition, once all the volunteer counts are collated, rigorous checks are made to identify and verify spurious records.

Conclusions and implications

Whereas it is realistic to expect that site-based SPA management will influence the actual numbers of birds wintering at a particular site, because of the influence of large-scale demographic processes it is unlikely that such an approach will have any discernible effect on the variance in these populations over time. Therefore, in an attempt to make the demands on site managers more realistic and perhaps improve their chances of success for migratory waterbirds in Great Britain, changes to the aims and objectives of conservation measures are recommended. As an alternative, we suggest a modification of the management requirements with regard to the imposition of a baseline population threshold (maintaining stable populations), whereby such a stipulation is removed from international agreements. Accordingly, managers will ultimately be more able to provide for the protection of target species into the future whilst being able to comply with the legal requirements of the Directive.

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