# Citizen science and Lepidoptera biodiversity change in Great Britain

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I certify that all material in this thesis which is not my own work has been identified and that any material that has previously been submitted and approved for the award of a degree by this or any other University has been acknowledged.

(Signature) ……………………………………….

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## Richard Fox

Frontispiece illustrations: Buff Arches *Habrosyne pyritoides* (left) and Comma *Polygonia c-album* (right) on Bramble by Richard Lewington. © Richard Lewington, reproduced with kind permission of the artist.

## Abstract

A considerable body of scientific evidence shows that the world is currently suffering a biodiversity crisis driven by anthropogenic factors such as land-use change, environmental pollution and climate change. Our knowledge of this crisis is incomplete, however, particularly when it comes to the most diverse multi-cellular organisms on the planet, the insects. Although there is evidence of decline in the abundance, distribution and biomass of many insect species, recent attempts to extrapolate these to global scales and encourage a policy response have been met with scepticism. More data are required, together with reliable methods to integrate and interpret them. In parallel, evidence-based conservation initiatives are urgently needed to address the biodiversity crisis.

Citizen science has great promise for gathering much-needed data on insect trends and for engaging the public in biodiversity conservation. Citizen science has undergone a rapid rise in popularity over the past two decades, increasing the capacity for cost-effective, spatially-extensive biodiversity monitoring, while also raising awareness and commitment to nature conservation among participating members of the public. However, citizen science approaches can also present challenges, such as reductions in data quality, constraints in sampling strategies and in the onward reuse of data.

In this thesis, citizen science monitoring of Great Britain's (GB) moths and butterflies is examined as a case study, assessing some of the benefits and limitations of increased participation and demonstrating applications of citizen science data in determining species trends, drivers of change and estimates of extinction risk.

Overall moth abundance has decreased in GB, probably mainly as a result of habitat degradation, while climate change has enabled the range expansion of some species (Chapter 2). Much remains to be learnt about other potential drivers of change, such as chemical pollution and artificial light at night (Chapter 2). I demonstrated the efficacy of citizen science by calculating GB distribution trends for 673 moth species for the first time, finding that 260 species had undergone statistically significant long-term declines compared with 160 that

had increased significantly (Chapter 3). The geographical patterns of change were consistent with expected responses to land-use, nutrient enrichment and climatic change (Chapter 3). I also utilised citizen-science derived monitoring data for 485 Lepidoptera species to investigate the impact of insect population variability on the assessment of Red List extinction risk using 10-year trends as specified by the International Union for Conservation of Nature procedure (Chapter 5). I concluded that for these taxa, strict use of 10-year trends produces Red List classifications that are unacceptably biased by the start year (Chapter 5).

In Chapter 4, I showed that mass-participation citizen science data obtained using a simple sampling protocol produced comparable estimates of butterfly species abundance to data collected through standardized monitoring undertaken by experienced volunteers. Resulting increases in participation, along with the associated benefits of public engagement and awareness raising, need not have a detrimental impact on the ability to detect abundance trends in common butterfly species. However, citizen science participation may affect the onward use of data, unless this is considered at the outset. I found that despite support in principle for open access to distribution records of butterflies and moths, most citizen scientists were much more cautious in practice, preferring to limit the spatial resolution of records, particularly of threatened species, and restrict commercial reuse of data (Chapter 6).

Overall, these results demonstrate the potential for citizen science, involving both expert volunteer naturalists and inexperienced members of the public, to address the global biodiversity knowledge gap through generating meaningful trend estimates for insect species and elucidating the drivers of change.

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## Author's declaration and contributions

The work contained in this thesis is my own. Chapters 1, 5 and 6 were written under the supervision of Dr Robert Wilson and Dr Ilya Maclean. Chapter 1 Extended Introduction was drafted by myself and reviewed by my supervisors, who offered advice and made suggestions for improvement. Collaborators on other chapters are listed below. Any further assistance has been acknowledged as appropriate at the end of each chapter.

This thesis for PhD by Publication is based on the following published papers:

Chapter 2
Fox R (2013) The decline of moths in Great Britain: a review of possible causes. *Insect Conservation and Diversity* **6**, 5-19.
I conceived the idea for this paper from an original suggestion by John Hopkins. I conducted the literature review, wrote the first draft of the manuscript and was responsible for subsequent editing.

Chapter 3
Fox R, Oliver TH, Harrower C, Parsons MS, Thomas CD & Roy DB (2014) Long-term changes to the frequency of occurrence of British moths are consistent with opposing and synergistic effects of climate and land-use changes. *Journal of Applied Ecology* **51**, 949-957.
I conceived the idea for this paper together with THO, CDT and DBR. I oversaw data collection and preparation, with input from MSP. Analyses were performed by THO and CH. I led the writing of the manuscript with THO. All authors contributed critically to drafts and I was responsible for subsequent editing.

Chapter 4
Dennis EB, Morgan BJT, Brereton TM, Roy DB & Fox R (2017) Using citizen science butterfly counts to predict species population trends. *Conservation Biology* **31**, 1350-1361.
I conceived the idea for this paper together with EBD and TMB. I oversaw data collection. Analyses were designed and performed by EBD and BJTM. EBD and

I led the writing of the manuscript and were responsible for subsequent editing. All authors contributed critically to drafts.

Chapter 5

Fox R, Harrower CA, Bell JR, Shortall CR, Middlebrook I & Wilson RJ (2019) Insect population trends and the IUCN Red List process. *Journal of Insect Conservation* **23**, 269–278.

I conceived the idea and designed the methodology with input from RJW. Moth population data were provided by JRB and CRS and analysed by CAH. I wrote the first draft of the manuscript and was responsible for subsequent editing, with all authors contributing significantly to revisions.

Chapter 6

Fox R, Bourn NAD, Dennis EB, Heafield RT, Maclean IMD & Wilson RJ (2019) Opinions of citizen scientists on open access to UK butterfly and moth occurrence data. *Biodiversity and Conservation* **28**, 3321–3341.

I conceived the idea and designed the methodology with input from NADB and RTH. I undertook analysis of the data following statistical and software guidance from EBD. I wrote the first draft of the manuscript and was responsible for subsequent editing, with all authors contributing significantly to revisions. RJW and IMDM offered advice and made suggestions for improvement.

## Acknowledgements

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I would like to thank all of the volunteer recorders, citizen scientists and colleagues involved in gathering, verifying and processing the UK butterfly and moth datasets on which most of the research presented in this thesis is based.

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# Chapter 1: Extended Introduction

## Global biodiversity change

Scientific research has provided compelling evidence of a global biodiversity crisis, sometimes characterised as the sixth mass extinction (Pimm *et al*. 1995; Barnosky *et al*. 2011; Dirzo *et al*. 2014). It is equally clear that this crisis is driven by human activities including habitat modification, overexploitation of species, nitrogen pollution, introduction of invasive species and anthropogenic climate change (Sala *et al*. 2000; Brook *et al*. 2008; Bellard *et al*. 2012; Newbold *et al*. 2015; Díaz *et al*. 2019).

The current biodiversity crisis is apparent in many demographic phenomena that can be estimated empirically. For example, the rate of species extinction in recent history is estimated to be 100-1000 times greater than the background rate measured across geological time (Pimm *et al*. 2014; Ceballos *et al*. 2015; De Vos *et al*. 2015). While relatively few modern extinctions have been documented (currently 872 species, comprising 750 animal species and 122 plants; IUCN 2018), a much larger number of species, estimated at c.9% (c.500,000 species) of terrestrial biodiversity (IPBES 2019), may already be destined for extinction as biotic communities transition to new equilibria following habitat loss or other environmental changes that have already occurred (Kuussaari *et al*. 2009; Halley *et al*. 2016). An overall increase in extinction risk has been shown for some vertebrate taxa using the Red List Index approach (e.g. amphibians and birds; Butchart *et al*. 2005), and estimated total vertebrate abundance decreased by 60% over the period 1970-2014 (WWF 2018). Approximately one third of land vertebrate species have decreased in abundance or range size (Ceballos *et al*. 2017). An abundance index based on studies of 452 invertebrate species (mostly insects) showed a 45% decline over 40 years (Dirzo *et al*. 2014), although the sample was heavily biased towards Europe and North America. Furthermore, significant decreases in biomass have been recorded in a variety of ecosystems (Shortall *et al*. 2009; Worm *et al*. 2009; Hallmann *et al*. 2017; Lister & Garcia 2018; Seibold *et al*. 2019).

However, not all species or populations are in decline (McGill *et al*. 2015; Dornelas *et al*. 2019), leading to counter-intuitive trends in some biodiversity metrics e.g. species richness. Meta-analyses of terrestrial plant and phytoplankton communities reported no net loss of local species richness despite high turnover in species identity and dominance (Vellend *et al*. 2013, Hillebrand *et al*. 2018), although these findings have been criticised (Cardinale *et al*. 2018). Herrera (2019) found increased flower visitation rates by insects over a 21-year study in undisturbed montane habitats in Spain, driven mainly by abundance increases of solitary bees. Even in taxa undergoing substantial overall decline in a particular region, there are species bucking the trend e.g. among bees (Powney *et al*. 2019) and moths (Boyes *et al*. 2019) in Great Britain (GB) and butterflies in the Netherlands (van Strien *et al*. 2019). The human impacts driving the global biodiversity crisis may also have positive effects on some species. Forest fragmentation can cause increased abundance in some vertebrates (Pfeifer *et al*. 2017), more intensive agricultural management benefits a minority of moths (Mangels *et al*. 2017), while urban-heat-island effects favour small-bodied invertebrates (Merckx *et al*. 2018). Anthropogenic climate change is causing regional range expansion of some species (Parmesan *et al*. 1999; Pearce-Higgins *et al*. 2017) and the introduction of non-native taxa not only increases their global distribution but also, in some cases, produces net gains in local biodiversity (Sax & Gaines 2003).

Because human environmental impacts generate winners and losers among species in a non-random way, another phenomenon of the biodiversity crisis is biotic homogenisation, whereby communities are increasingly similar and often dominated by a relatively small number of generalist species that are well-adapted to highly-modified landscapes (McKinney & Lockwood 1999; Smart *et al*. 2006; Le Viol *et al*. 2012; Carvalheiro *et al*. 2013; Gossner *et al*. 2016).

Overall, these changes threaten the resilience of ecological communities and the provision of ecosystem services upon which the human race depends (Chapin *et al*. 2000; Cardinale *et al*. 2012; Bernstein 2014; Oliver *et al*. 2015a). This holds irrespective of the ongoing debate regarding the relative importance for ecosystem functioning of high species diversity (Isbell *et al*. 2011; Tilman *et al*. 2014; but see Pillai & Gouhier 2019) versus high abundance of common species (Gaston & Fuller 2008; Kleijn *et al*. 2015), as there is ample evidence of decline in both.

Yet, despite political agreements to address global biodiversity loss (e.g. the Convention on Biological Diversity), drivers of change continue to intensify and biodiversity continues to decline (Butchart *et al*. 2010; Tittensor *et al*. 2014; Díaz *et al*. 2019). Unless radical steps are taken to address human pressures on biodiversity, projections indicate further severe losses (Pereira *et al*. 2010; Maclean & Wilson 2011; Visconti *et al*. 2016; Warren *et al*. 2018).

Insects represent a key gap in our understanding of biodiversity change. In this thesis, I focus on Lepidoptera (moths and butterflies) in GB to assess evidence of insect biodiversity change. More specifically, I seek to gauge the potential for citizen science to provide robust data on insect trends. Awareness of the problems of biodiversity loss is increasing globally (Díaz *et al*. 2019) and, in GB, volunteer time devoted to conservation organisations has increased by 46% this century (Hayhow *et al*. 2019). I investigate the scope to which this public engagement can be harnessed to document and understand change in GB Lepidoptera populations, addressing several factors that may influence the effectiveness of such an approach.

### The insect information gap

Insects are the most speciose eukaryotic organisms on Earth (Mora *et al*. 2011) and are essential to the functioning of many terrestrial and freshwater ecosystems (Wilson 1987; Collen *et al*. 2012; Yang & Gratton 2014). They have numerous economically significant impacts on humans, including positive ecosystem services such as pollination (Losey & Vaughan 2006; Gallai *et al*. 2009) and negative interactions e.g. as disease vectors and crop pests (Bradshaw *et al*. 2016). Despite their ecological and economic importance, insects are poorly represented in assessments of extinction risk and biodiversity change (Dunn 2005; Stuart *et al*. 2010; Cardoso *et al*. 2011a,b). For example, global extinction risk has been assessed for 100% bird and mammal species and 67% of all vertebrate taxa, but only 0.8% of described insect species (Eisenhauer *et al*. 2019). Empirical estimates in well-studied regions suggest that proportions of threatened species and rates of decline in insects exceed or are similar to those of vertebrate taxa (McKinney 1999; Thomas *et al*. 2004; Sánchez-Bayo & Wyckhuys 2019), suggesting that real rates of species extinction globally are much higher than previous, vertebrate-based estimates (Régnier *et al*. 2015). Indeed, given the specialized niches and co-dependence of many insect species, extinction rates may be expected to greatly exceed those of other taxa (Koh *et al*. 2004; Dunn 2005; Fonseca 2009).

The under-representation of insects in considerations of biodiversity change stems from insufficient knowledge of insect taxonomy, ecology and biogeography, reinforced by societal preferences (Diniz-Filho *et al*. 2010; Cardoso *et al*. 2011a; Troudet *et al*. 2017). Invertebrates are greatly under-represented in conservation biology research compared with vertebrates or plants (Clark & May 2002; Deikumah *et al*. 2014). Even among invertebrates, insects are under-represented relative to their species richness in scientific publications (Di Marco *et al*. 2017). For the vast majority of insect species across most of the world, occurrence and population data from which to compile trends in insect biodiversity are non-existent (IPBES 2019; Wagner 2020).

While this information gap remains a major impediment to large-scale assessment of insect biodiversity change, substantial progress has been made recently in developed nations for certain insect groups. These include continental or national-scale assessments of butterflies (van Swaay *et al*. 2011; Breed *et al*. 2013; Maes *et al*. 2019), bees (Cameron *et al*. 2011; Nieto *et al*. 2014; Kerr *et al*. 2015), moths (Conrad *et al*. 2006; Groenendijk & Ellis 2011), ladybirds (Harmon *et al*. 2007; Roy *et al*. 2012), hoverflies (Powney *et al*. 2019) and dragonflies (Kalkman *et al*. 2010; Termaat *et al*. 2019). In addition, a partial global Red List assessment has been undertaken for Odonata (Clausnitzer *et al*. 2009), using a sample of 1500 species, and the same approach has been advocated for butterflies (Lewis & Senior 2011).

Nevertheless, huge gaps remain for other insect taxa, particularly those that are species-rich or perceived to be less appealing (Sumner *et al*. 2018), and for biodiversity as a whole in many developing countries (Collen *et al*. 2008; Stephenson *et al*. 2016; Barlow *et al*. 2018).

## Insect Armageddon

Despite the general paucity of data, several recent papers about insect declines have achieved a high profile in the media and stimulated vigorous debate within the scientific community. Hallmann *et al*. (2017) estimated a 77% reduction in the biomass of flying insects over 27 years in German nature reserves, with an even greater loss (82%) in mid-summer when biomass values peak. Lister & Garcia (2018) reported dramatic declines in biomass and abundance across all the main insect groups in a protected rainforest in Puerto Rico. For example, arthropod biomass in ground-level sticky traps decreased by 36-60 times (depending on the season) between 1976-1977 and 2011-2013. Despite many previous studies documenting insect population decline, these two studies generated major media coverage with much hyperbole and the invention of phrases such as “ecological armageddon”, “insect apocalypse” and “insectageddon”. A global review of insect declines (Sanchez-Bayo & Wyckhuys 2019) coincided with the media zeitgeist and also received huge publicity. This review did nothing to temper exaggerated reporting, with the authors suggesting that insects could become extinct within decades: “The conclusion is clear: unless we change our ways of producing food, insects as a whole will go down the path of extinction in a few decades” (Sanchez-Bayo & Wyckhuys 2019).

These studies and the associated media coverage provoked a considerable response from the scientific community. While the evidence for overall declines in insect biodiversity is clear (Wagner 2020) and raising public awareness vital, potential problems with the analyses and extrapolation of local results to the global scale leading to overstated claims have been widely criticised (Leather 2018; Saunders 2019; Simmons *et al*. 2019; Thomas *et al*. 2019). Researchers have called instead for investment in more rigorous and spatially extensive monitoring, greater sharing of ecological data and the deployment of new technologies to provide better estimates of global insect biodiversity change (Montgomery *et al*. 2020; Saunders *et al*. 2020). Others have pointed out that, even excluding the recent media hype, there is a sufficient scientific basis to be deeply concerned about insect declines and to start to formulate policy responses (Forister *et al*. 2019; Habel *et al*. 2019a; Harvey *et al*. 2020).

## Citizen science and insect biodiversity assessment

Citizen science has huge potential to help fill the data shortfalls that hinder comprehensive biodiversity assessments of insects (Danielsen *et al*. 2014; Theobald *et al*. 2015; McKinley *et al*. 2017; Pocock *et al*. 2018). Sumner *et al*. (2019), for example, showed that a two-week UK citizen science project could generate comparable spatial coverage for social wasp species as four decades of recording by expert amateurs. Furthermore, participation in citizen science also engenders increased understanding of and engagement with conservation (Jordan *et al*. 2011; Haywood *et al*. 2016; Domroese & Johnson 2017; Lewandowski & Oberhauser 2017) and may benefit mental wellbeing (Coventry *et al*. 2019). Although ‘citizen science’ is a recently coined term, it is nothing new; non-professional scientists and members of the public have been involved in gathering data and undertaking research in many branches of science for centuries (Miller-Rushing *et al*. 2012; Kobori *et al*. 2016). However, there has been a rapid growth in the number of citizen science projects, participants and research publications in the past two decades, particularly in ecology (Silvertown 2009; Follett & Strezov 2015; Pocock *et al*. 2017).

Citizen science already makes a considerable contribution to knowledge of the status, distribution, abundance and trends of biodiversity (Dickinson *et al*. 2012; Chandler *et al*. 2017). This is particularly the case for birds (e.g. Gibbons *et al*. 2007; VanDerWal *et al*. 2013; Sullivan *et al*. 2017; Lehikoinen *et al*. 2019), but also applies to insects (e.g. Biesmeijer *et al*. 2006; van Swaay *et al*. 2008; Soroye *et al*. 2018) and other taxa.

Nowhere is this more evident than in GB. Thanks to a long tradition of citizen science (Pocock *et al*. 2015), GB biodiversity is probably the most thoroughly monitored in the world (Burns *et al*. 2018). Certain charismatic taxa (e.g. birds, butterflies) are the focus of long-term standardised monitoring schemes that utilise repeatable, distance sampling techniques and skilled volunteer observers to produce count data from which population estimates can be derived (Freeman *et al*. 2007; Roy *et al*. 2007; Dennis *et al*. 2016).

In addition to such ‘gold standard’ monitoring, opportunistic sightings (occurrence records) of many GB taxa are contributed by citizen scientists through national recording schemes and local environmental records centres (Thomas 2005; Powney & Isaac 2015). However, species records made by citizen scientists and without standardised sampling protocols are subject to data quality problems (Kosmala *et al*. 2016) and bias resulting from uneven sampling (Boakes *et al*. 2010; Isaac & Pocock 2015). In order to account for sampling bias in estimates of species distribution change, a range of statistical approaches has been developed (Pardo *et al*. 2013; Bird *et al*. 2014; Isaac *et al*. 2014; Dennis *et al*. 2017a). Through the application of such techniques, long-term distribution changes have been assessed for thousands of GB taxa (Hickling *et al*. 2006; Burns *et al*. 2018; Outhwaite *et al*. 2019) and the drivers of change examined (Burns *et al*. 2016). Relatively minor adjustments to the collection of citizen science data could also yield significant improvements in the accuracy of scientific outputs (Altwegg & Nichols 2019; Callaghan *et al*. 2019; Kelling *et al*. 2019).

### Lepidoptera citizen science in GB

Citizen science, broadly taken to mean the involvement of non-professionals in the generation of scientific knowledge (Strasser *et al*. 2019), encompasses both the long-standing participation of amateur (but often expert) naturalists in the recording of fauna and flora (see Asher *et al*. 2001 and Randle *et al*. 2019 for histories of GB Lepidoptera recording) and recent projects designed to engage the wider public (Pocock *et al*. 2017). The research in this thesis draws upon Lepidoptera data from five GB citizen science schemes: Butterflies for the New Millennium, the National Moth Recording Scheme, the UK Butterfly Monitoring Scheme, the Rothamsted Insect Survey and Big Butterfly Count. Other projects that collect Lepidoptera records, e.g. the Garden Moth Scheme (Wilson *et al*. 2015), Moth Night (www.mothnight.info), Garden Butterfly Survey (www.butterfly-conservation.org) and Garden BirdWatch (www.bto.org), are not considered here.

#### *Butterflies for the New Millennium*

The distribution recording scheme for butterflies, Butterflies for the New Millennium (BNM), is operated by Butterfly Conservation and collates occurrence records across the UK. Records (unique combinations of species x recorder x location x date) are opportunistic and sampling is neither standardised nor systematic. Records can be of any life-cycle stage on any date and at any terrestrial location. The scheme began in 1995 as a five-year survey to map current species distributions and measure change over time for a butterfly atlas. This initial phase gathered 1.6 million records from an estimated 10,000 citizen scientists and covered 98.7% of the 10km x 10km grid squares across the study area. These data and historical records (mainly from Heath *et al*. 1984) were used to produce the planned atlas (Asher *et al*. 2001) and assessments of change (e.g. Warren *et al*. 2001; Thomas *et al*. 2004).

Back-to-back five-year BNM surveys have continued since then, with increasing recording effort, and the scheme currently holds 14.0 million butterfly records (13.6 million for GB), the earliest dating from 1690 (Fox *et al*. 2015). Records are gathered and verified by a network of expert volunteer ‘County Recorders’, prior to being collated into the BNM database.

### *National Moth Recording Scheme*

The National Moth Recording Scheme (NMRS) was launched by Butterfly Conservation in 2007 to create a UK database of moth occurrence records to underpin conservation (Fox *et al*. 2011a). Initially focussed on c.900 species of macro-moths, the scheme amassed 25.3 million records covering the period 1741-2016 (Randle *et al*. 2019). Recently, the NMRS has widened to include micro-moths. NMRS recording is largely unstructured and opportunistic and County Recorders collate and verify local datasets, which are then merged into a single database. Most records are of nocturnal adult moths attracted to light-traps, but sightings of any life-cycle stage at any time of day can be contributed. Thus, in contrast to the Rothamsted Insect Survey, the NMRS gathers data on all macro-moths, not just nocturnal species that are attracted to light.

Recording has increased greatly over time. For example, the NMRS contains c.680,000 pre-1970 records, representing 73% of 10km x 10km grid squares in GB, but has 17.9 million records from 97% of 10km grid squares for 2000-2016 (Fig. 1.1) (Randle *et al*. 2019). The number of participants is unknown, because of variation in the way that recorders’ names are collated by County Recorders, but is estimated at 5,000-10,000.

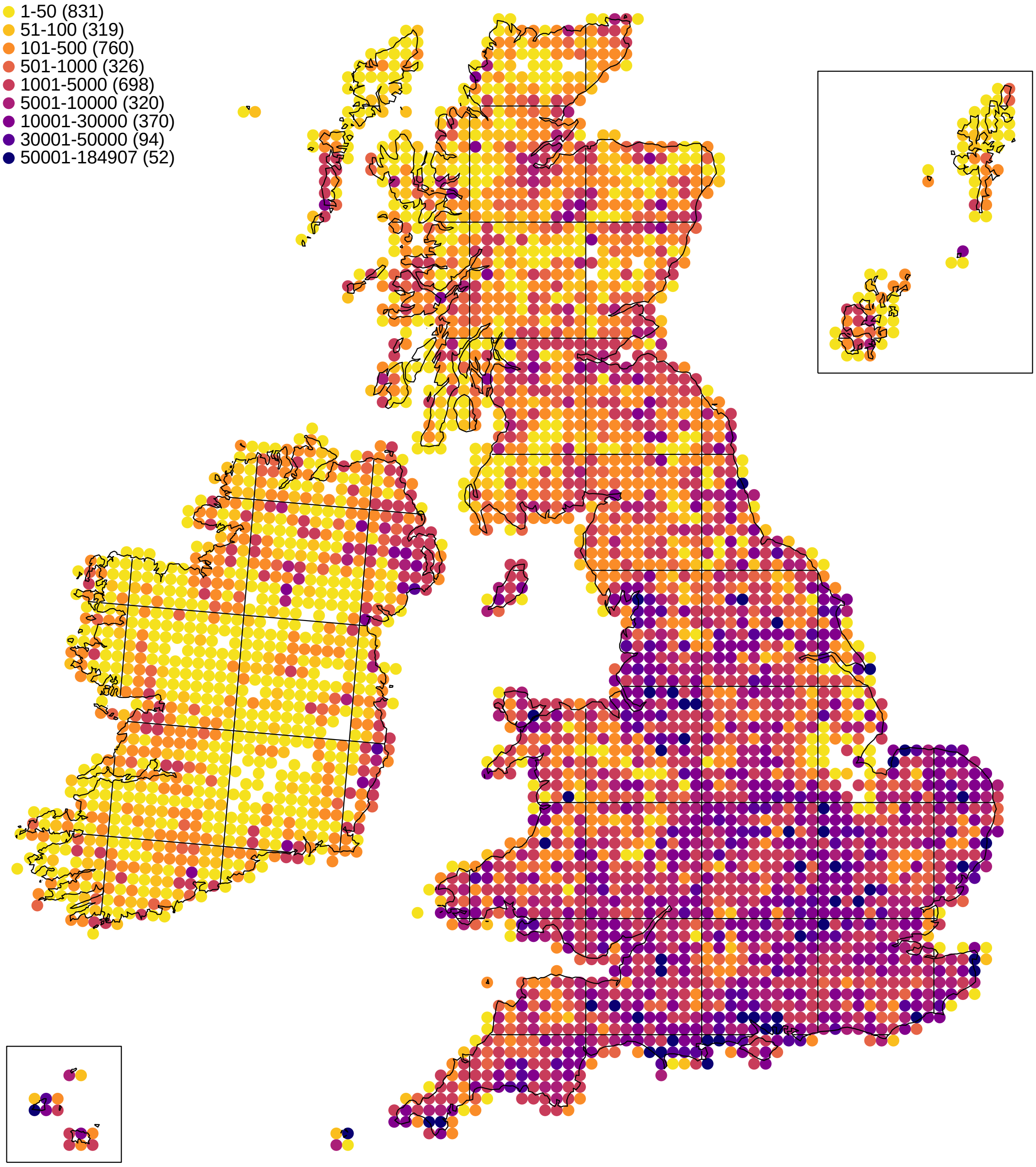

**Fig. 1.1** NMRS recording coverage illustrated by the number of macro-moth records per 10km grid square (2000-2016). The data for GB, Isle of Man and Channel Islands comes from the NMRS. Data for the Republic of Ireland comes from MothsIreland. Both the NMRS and MothsIreland contain records for Northern Ireland, so the map shows data from both schemes. The number of 10km squares in each numerical banding is given in the key. This map also appears in Randle *et al*. (2019).

### *UK Butterfly Monitoring Scheme*

A standardised method for measuring change in the relative abundance of butterflies along fixed-route, 5m wide transects (Pollard 1977) was rolled out as a UK-wide scheme in 1976 (Pollard & Yates 1993). Transect locations are selected by volunteer recorders, who then undertake weekly counts in good weather from April-September each year. Some sites have been monitored over many years, but there is turnover of locations due to volunteer availability. While the monitoring is standardised, enabling counts to be combined and compared across sites and years, transects are biased towards high biodiversity sites managed (at least partly) for nature conservation. The skewed distribution of transects ensures sufficient coverage of rare species to enable annual population indices to be produced for almost all UK butterflies, but is not representative of the wider landscape. To address this, a reduced-effort transect methodology was developed (Roy *et al*. 2007) and applied to a stratified random sample of 1km x 1km grid squares for the Wider Countryside Butterfly Survey (Brereton *et al*. 2011a).

The UK Butterfly Monitoring Scheme (UKBMS) is run by the UK Centre for Ecology & Hydrology, Butterfly Conservation and British Trust for Ornithology and incorporates data annually from >1,700 transects, c.800 Wider Countryside Butterfly Survey squares and c.300 sites where timed counts are carried out (Fig. 1.2). Thus, in 2018, approximately 2,500 UKBMS citizen scientists walked some 92,000km counting butterflies at 2,868 sites (Brereton *et al*. 2019).

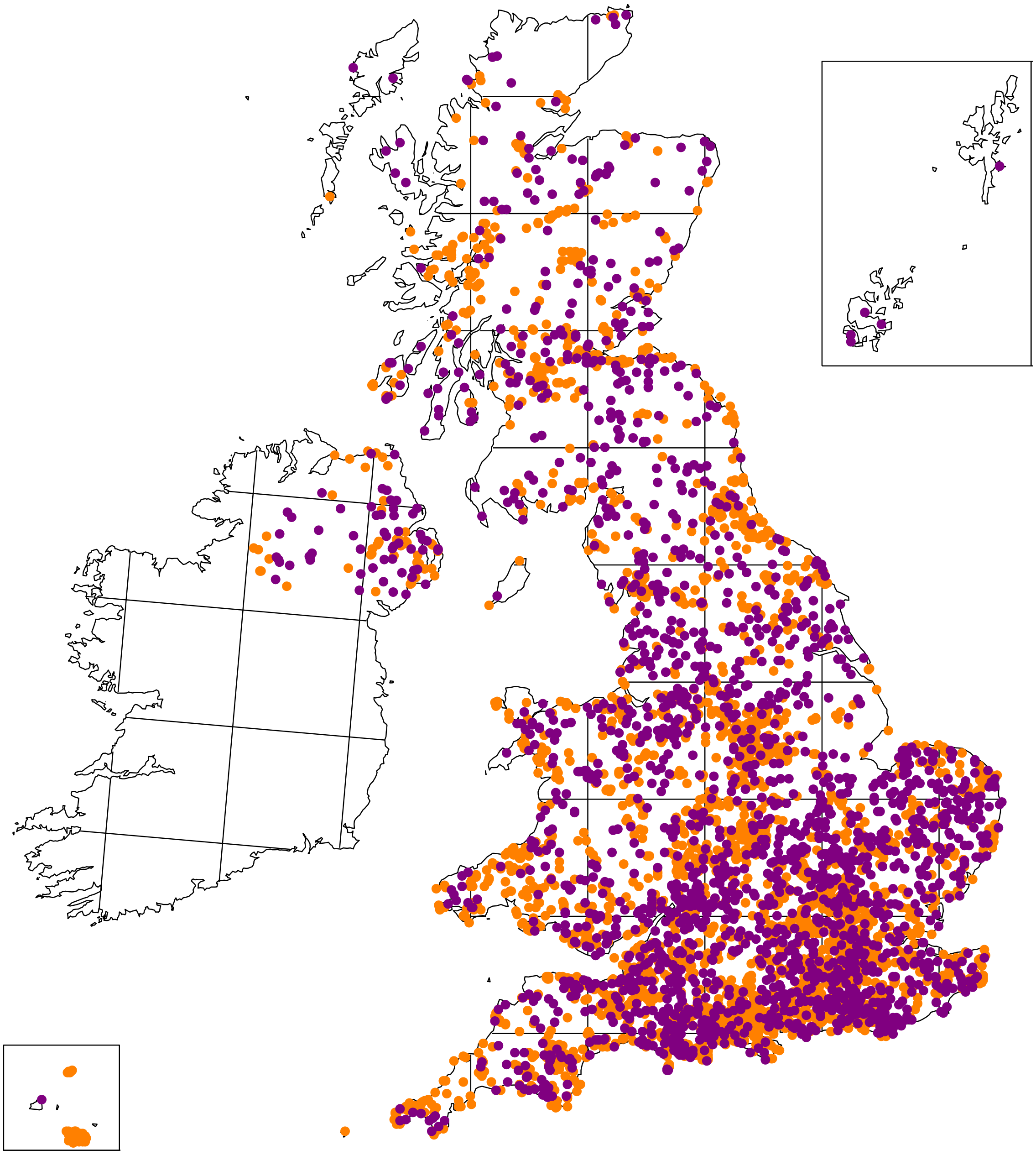

**Fig. 1.2** Locations of UKBMS sites (1976-2018). Purple dots show transects, orange dots show Wider Countryside Butterfly Survey squares.

*Rothamsted Insect Survey*

In the 1960s, Rothamsted Research set up two separate UK networks to monitor insects, particularly those of interest to the agricultural sector. These are managed collectively as the Rothamsted Insect Survey (RIS) (Storkey *et al*. 2016). One comprises continuously-running suction-traps, used mainly to monitor aphid populations (Bell *et al*. 2015), although it has also been used to assess trends in aerial insect biomass (Shortall *et al*. 2009). The other is a network of standardised, automated light-traps, which operate nightly and are used to monitor the relative abundance of nocturnal macro-moths (Woiwod *et al*. 2005; Conrad *et al*. 2007). RIS light-traps have operated at c.540 sites for various durations between 1968-2018, with c.80 traps active each year in recent times. The first RIS light-trap was operated at Rothamsted Research in the 1930s and 1940s and the fundamental design of the trap and light source has remained unchanged since then (Williams 1948).

The RIS light-trap network has less citizen science involvement than the other schemes detailed here. For much of its history, the catch from most traps was counted by professional entomologists, although some traps have always been monitored by volunteers and nowadays most are done by expert citizen scientists.

### *Big Butterfly Count*

In contrast to the previously discussed schemes, which all have records extending back to at least the mid-1970s, Butterfly Conservation's Big Butterfly Count has collected count data for selected, widespread butterflies and diurnal macro-moths during a short survey window (usually three weeks) each summer since 2010 (Dennis *et al*. 2017b). Big Butterfly Count also differs from the other schemes in that an explicit aim is to engage new audiences, outside of the existing community of biological recording volunteers. To minimize barriers to participation, a simple sampling protocol is used, comprising a 15-minute count of the (currently) 19 target species. There is no other standardisation of sampling effort and no verification of records before analysis.

Participation rates are high compared to most citizen science biodiversity projects. An estimated 113,000 people carried out 116,000 counts during Big Butterfly Count 2019. Most counts take place in gardens (mean 65% p.a. 2011-2014) and are biased towards urban landscapes (Dennis *et al*. 2017b); 1km squares with Big Butterfly Counts contained a mean of 33.4% urban landcover, which contrasts with a mean of 9.8% for squares sampled by the UKBMS.

## Lepidoptera change in GB

Thanks to long-term citizen science, the changing status of GB's Lepidoptera fauna is well documented.

Population and distribution trends of butterfly species have been regularly assessed since the 1970s, repeatedly demonstrating decreases for most species (Heath *et al*. 1984; Asher *et al*. 2001; Fox *et al*. 2015) and greater overall declines than equivalent trends for mammals, birds and vascular plants (Thomas *et al*. 2004; Hayhow *et al*. 2019). The most recent assessment, using UKBMS and BNM data, found that 76% of butterfly species had decreased in either abundance or occurrence or both over the period 1976-2014, while 47% increased in one or both measures (Fox *et al*. 2015). The GB Red List of butterflies categorises 31% of species as threatened (Fox *et al*. 2011b).

Multi-species population indicators using UKBMS data have been adopted by Government for environmental monitoring (Brereton *et al*. 2011b). These show long-term (1976-2018) statistically significant decreases of 68% for habitat specialist butterflies and 30% for wider countryside species (Defra 2019). Analysis of BNM distribution trends also showed habitat specialists faring worse than generalists (Warren *et al*. 2001), and this pattern has been found widely in butterflies (Öckinger *et al*. 2010; Eskildsen *et al*. 2015; Habel *et al*. 2019b) and other taxa (Clavel *et al*. 2011; MacLean & Beissinger 2017; Platts *et al*. 2019).

Declines of habitat specialist butterflies in GB are long-term, driven by land-use changes that commenced in the first half of the 20th century, such as agricultural intensification and reductions in woodland management. Severe decreases of species such as Heath Fritillary *Melitaea athalia* (Warren *et al*. 1984), Marsh Fritillary *Euphydryas aurinia* (Warren 1994) and Large Blue *Maculinea arion* (Thomas 1980) were documented decades ago and conservation programmes initiated. More recently, abundance declines of some widespread butterflies (e.g. Wall *Lasiommata megera*, Small Tortoiseshell *Aglais urticae*) have become evident in GB (Gripenberg *et al*. 2011; Fox *et al*. 2015) and neighbouring countries (Van Dyck *et al*. 2009; Van Dyck *et al*. 2015).

Data with which to evaluate biodiversity change among GB moths, while good compared to most insect taxa, are less comprehensive than for butterflies. National-scale assessments of moth biodiversity have been undertaken however, albeit based on subsets of species (Fox *et al*. 2013; Chapter 3). Sixty-five moth species were extirpated from GB over the period 1900-2009, while

112 species became established (Parsons 2010). Long-term population trends have been derived for 337 relatively widespread macro-moths using RIS count data (Conrad *et al*. 2004). Two-thirds of these had negative trends and the overall abundance of GB macro-moths decreased by 31% over 35 years (1968-2004) (Conrad *et al*. 2006). The decrease in overall abundance was even more marked in southern GB (44% decline), while there was no significant change in the northern half of GB (Conrad *et al*. 2006). A more recent analysis of macro-moth abundance from RIS traps in Scotland, however, showed a significant decrease of 20% for 1975-2014 (Dennis *et al*. 2019).

RIS data for the most rapidly declining widespread moth species also contribute, with population monitoring of a small number of rare moths, to an UK Government indicator of priority species (Eaton *et al*. 2015). Of the four taxonomic groups in this indicator, moths (76 species) showed the greatest abundance decline, with a 2016 index value that was only 14% of the baseline 1970 value. Butterflies (23 species) also experienced a strong decline, with a 2016 index value that was 17% of its start value in 1976, while the bird index (104 species) showed no overall change and the mammal index (11 species) increased (Burns *et al*. 2019).

Macgregor *et al*. (2019a) recently estimated biomass change of GB moths using RIS data. While the effect size of their trend suggested a 32% decrease over the period 1983-2017, this was far less severe than the 77% decrease in flying insect biomass reported in Germany over a similar period (Hallmann *et al*. 2017). Surprisingly, however, Macgregor *et al*. (2019a) found a major increase in GB moth biomass during 1967-1982, meaning that biomass levels were still much higher in 2017 than they had been in 1967.

In addition to abundance and biomass change, some GB moth species have shown large contractions or expansions of range (Randle *et al*. 2019). In a study of NMRS data for 673 moth species, Fox *et al*. (2014; Chapter 3 of this thesis) found that 39% had significant negative trends in frequency of occurrence (1970-2010), compared with 24% that had significant positive trends. Recently, an occupancy modelling approach generated long-term (1970-

2016) trends for 390 species at a finer spatial scale (Randle *et al*. 2019); 121 species (31%) decreased significantly, while 148 (38%) increased significantly.

## Drivers of Lepidoptera biodiversity change

Utilising citizen science data in GB and elsewhere (particularly in western Europe and North America e.g. Schmucki *et al*. 2016; Wepprich *et al*. 2019), as well as experimental results, considerable progress has been made in understanding the causative environmental drivers of Lepidoptera biodiversity change (discussed further in Chapters 2 and 3).

### *Land use*

Land-use change, including both the conversion of semi-natural habitats to highly modified landscapes and major increases/decreases in management intensity, is thought to have been a key driver over recent decades (Thomas 1995; Warren *et al*. 2001; Bubová *et al*. 2015; Thomas 2016). The impact of intensive agriculture has been particularly harmful (Burns *et al*. 2016; Hayhow *et al*. 2019). Higher diversity and abundance of butterflies and moths are typically found in less intensively managed agricultural habitats (Mangels *et al*. 2017; Habel *et al*. 2019c) and reduced management (e.g. through agri-environment schemes) often leads to increased numbers and species richness (Fuentes-Montemayor *et al*. 2011; Zingg *et al*. 2019). For example, reduced frequency and intensity of hedgerow cutting in GB agricultural landscapes benefitted Lepidoptera communities (Staley *et al*. 2016; Staley *et al*. 2018; Froidevaux *et al*. 2019).

However, reduced management can also drive Lepidoptera decline. Abandonment of low-productivity grasslands in Europe, with subsequent succession to woodland, has impacted negatively on specialist butterflies and moths of open habitats (Nilsson *et al*. 2013; Herrando *et al*. 2016; Ubach *et al*. 2019), as has the cessation of traditional woodland management (e.g. coppicing) (Warren & Key 1991; Fartmann *et al*. 2013; Thomas *et al*. 2015). On the other hand, many moth species are associated with woodland habitat and should benefit from afforestation and reduced management (Merckx 2015).

Urbanisation is linked to reductions in Lepidoptera abundance and species richness, particularly the loss of habitat specialists (Deguines *et al*. 2016; Ramírez-Restrepo & MacGregor-Fors 2017; Merckx & Van Dyck 2019). Furthermore, in an assessment of population change for GB butterfly species (1995-2014) in urban versus rural UKBMS locations, trends were more negative in urban areas for 25 of 28 species and a composite index of all species showed a significantly greater decrease for urban than for rural areas (-69% for urban compared with -45% for rural) (Dennis *et al*. 2017c).

Habitat loss decreases the size of remaining patches and increases their isolation. Both can increase the risk of local extirpation of Lepidoptera species through reduced population size (leading to greater extinction rates) and decreased dispersal (leading to lower colonisation rates) (Thomas 2000; Öckinger *et al*. 2010).

In contrast, land management to enhance biodiversity can benefit Lepidoptera populations. This is exemplified by the successful reintroduction of the Large Blue to GB (Thomas *et al*. 2009), but also in many other examples where threatened species have been the focus of conservation action e.g. New Forest Burnet *Zygaena viciae* (Young & Barbour 2004), High Brown Fritillary *Argynnis adippe* (Ellis *et al*. 2019). Although not immune from biodiversity declines, protected areas of high-quality habitat maintain higher abundance and species richness of butterflies than the surrounding landscape (Gillingham *et al*. 2015; Rada *et al*. 2018) and agri-environment schemes have benefitted some declining species (e.g. Brereton *et al*. 2008).

### *Environmental pollution*

Disentangling the impact of pesticides from other aspects of intensive management is difficult, particularly given the lack of ecotoxicological data for insecticides on non-target Lepidoptera (Pisa *et al*. 2015; Braak *et al*. 2018) and the potential for direct (Russell & Schultz 2010; Stark *et al*. 2012) and indirect effects of herbicides via impacts on larval hostplants (Prosser *et al*. 2016; Belsky & Joshi 2018). Nevertheless, the overall impact of systemic insecticides is thought to be substantial (Chagnon *et al*. 2015), routes of exposure via nectar, pollen and tissues of wild plants have been demonstrated (Botías *et al*.

2015; Botías *et al*. 2016; Basley & Goulson 2018) and correlative studies using citizen science data suggest negative effects on butterfly populations (Gilburn *et al*. 2015; Muratet & Fontaine 2015; Forister *et al*. 2016). A short-term field experiment found negative effects of insecticide application on moth caterpillar abundance in field margins, but no effect of herbicide treatment (Hahn *et al*. 2015).

Nutrient enrichment is expected to affect insect herbivores via changes to the chemistry, structure and composition of plant communities (Nijssen *et al*. 2017; Stevens *et al*. 2018). Several studies have found positive correlations between Lepidoptera species trends and the Ellenberg nitrogen indicator values of their larval hostplants (where plants preferring fertile soils have higher Ellenberg nitrogen scores) (Öckinger *et al*. 2006; Betzholtz *et al*. 2013; Fox *et al*. 2014; Pöyry *et al*. 2017; WallisDeVries & van Swaay 2017). Kurze *et al*. (2018) recorded increased larval mortality in all six study species of grassland Lepidoptera when nitrogen fertilizer was applied to hostplants at rates typically used in agriculture. In a similar study, positive responses were found in two butterfly species to nitrogen fertilization of their nitrophilous hostplant (Kurze *et al*. 2017).

Reduced air pollution in GB may also be indirectly driving some species trends. Moths with larvae that feed on lichens, e.g. Dingy Footman *Eilema griseola* and Marbled Green *Nyctobrya muralis*, have fared well against a background of general decline (Conrad *et al*. 2004; Randle *et al*. 2019). Using citizen science data, Pescott *et al*. (2015) linked this to the recovery of lichen populations following air quality improvements, particularly reduction in sulphur dioxide pollution.

Urbanisation is accompanied by large increases in artificial light at night. The global extent and intensity of artificial light are both increasing at c.2% per year and 83% of the human population now lives under light-polluted skies (Gaston 2018). Artificial light can alter invertebrate community composition (Manfrin *et al*. 2017), drive population change (Bennie *et al*. 2018) and affect insect behaviour (Owens & Lewis 2018). As yet, there is no evidence for a direct causative link with population change in any Lepidoptera species, although many moths are attracted to light (e.g. Somers-Yeates *et al*. 2013) and nocturnal moths that exhibit positive phototaxis decreased more over 30 years in the Netherlands than diurnal species or moths that are not attracted to light (Van Langevelde *et al*. 2018). Artificial light can disrupt pheromone production in Cabbage Moth *Mamestra brassicae* (Van Geffen *et al*. 2015a), reduce mating in Winter Moth *Operophtera brumata* (Van Geffen *et al*. 2015b), decrease larval growth in Rustic Shoulder-knot *Apamea sordens* (Grenis & Murphy 2019) and inhibit feeding in adult moths (Van Langevelde *et al*. 2017). It has also been shown to alter nocturnal pollination by moths and other insects (Knop *et al*. 2017, Macgregor *et al*. 2019b). While research has focussed on nocturnal insects, artificial light could influence populations of diurnal Lepidoptera directly (e.g. through impacts on nocturnal larvae) and indirectly via effects on the growth and phenology of hostplants (Bennie *et al*. 2016; ffrench-Constant *et al*. 2016).

*Climate change*

Substantial effects of climate change on Lepidoptera populations are evident in GB and elsewhere (e.g. Parmesan *et al*. 1999; Parmesan 2006; Chen *et al*. 2011; Molina-Martínez *et al*. 2016). Citizen science data from the BNM and UKBMS have shown climate-related shifts in distribution and abundance for butterflies in GB and Europe (Warren *et al*. 2001; Hill *et al*. 2002; Devictor *et al*. 2012). Drawing on BNM and NMRS data, Mason *et al*. (2015) showed that the distributions of butterflies and moths with northern range margins in GB have expanded polewards at an increasing rate since the 1960s. Risk assessments predict that 46% of 52 butterfly species and >60% of 422 moths could increase in overall extent in GB due to climate change this century (Thomas *et al*. 2011; Pearce-Higgins *et al*. 2017).

Other studies, however, have found evidence of climate-driven range contraction (e.g. Thomas *et al*. 2006; Breed *et al*. 2013), although microclimatic buffering may ameliorate these effects (Suggitt *et al*. 2018), and negative impacts on Lepidoptera population growth (Conrad *et al*. 2002; Palmer *et al*. 2017). Analysing RIS data, Martay *et al*. (2017) implicated climate change as a major driver in the population declines of some moth species and in the overall abundance decline of macro-moths. In addition, negative impacts on butterfly abundance of extreme climatic events, which are expected to increase with climate change, have been demonstrated using UKBMS data. Oliver *et al*. (2015b) predicted substantial long-term abundance declines of six drought-sensitive butterflies in response to the increasing frequency of summer drought, while extreme winter warmth exerted detrimental population effects in 21 of 41 butterfly species (versus only two species with positive effects) (McDermott Long *et al*. 2017).

Changes to phenology are common biotic responses to climate change (Cohen *et al*. 2018), including among butterflies and moths (Roy & Sparks 2000; Stefanescu *et al*. 2003; Kearney *et al*. 2010), raising concerns about temporal mismatches in ecological interactions (Thackeray *et al*. 2016; Burgess *et al*. 2018). Using GB citizen science data for 130 Lepidoptera species, Macgregor *et al*. (2019c) showed that phenological advance was associated with increased population growth but only in multivoltine species; flight periods of univoltine species did advance significantly over a 20-year period but there was no clear relationship with abundance trends. Patterns of voltinism are also changing in response to climate change, with increased incidence of multiple broods per year (Altermatt 2010; Pöyry *et al*. 2011). The demographic consequences of such changes are not yet understood, although a third generation 'development trap' has been proposed as a cause for the decline of the Wall butterfly (Van Dyck *et al*. 2015).

### *Interactions between drivers of change*

Drivers can act synergistically to amplify impacts on biodiversity (Brook *et al*. 2008) or in opposition to reduce responses. Interactions of land-use change and climate change have been investigated by numerous studies (Mantyka-pringle *et al*. 2012; Guo *et al*. 2018; Newbold *et al*. 2019; Northrup *et al*. 2019). In fragmented GB landscapes, the ability of species to track climate change by shifting their distributions is limited by habitat availability (Mair *et al*. 2014; Platts *et al*. 2019). The range expansion of Speckled Wood *Pararge aegeria* was slower in landscapes with less woodland (Hill *et al*. 2001), while the spread of Silver-spotted Skipper *Hesperia comma* was facilitated by conservation management of protected areas (Lawson *et al*. 2014).

Climate change and nitrogen deposition can also act in synergy to increase vegetation growth, reducing the availability of warm micro-climates for thermally-constrained species. As yet there is limited evidence for this as a driver of population change in Lepidoptera, although it has been implicated in the decline of European butterflies that overwinter in the egg or larval stage (WallisDeVries & van Swaay 2006) and specific species such as the Wall (Klop *et al*. 2015) and High Brown Fritillary (Ellis *et al*. 2019). De Sassi *et al*. (2012) found independent and synergistic effects of climate change and nitrogen deposition on Lepidoptera community composition and biomass, mediated through changes in plant species dominance and quality.

## Overview of the rationale of this thesis

Although Lepidoptera biodiversity recording through citizen science is well developed in GB, many gaps in our knowledge remain. My thesis aims to address two overarching areas regarding the efficacy of citizen science in biodiversity conservation:

1. Increasing citizen science participation while ensuring data validity.
2. Application of citizen science data to biodiversity conservation.

Despite statistical improvements (Dennis *et al*. 2013; Isaac *et al*. 2014), increased citizen science recording effort is desirable to increase the quantity and quality of species trends. For example, even with the high levels of UK recording, Outhwaite *et al*. (2019) had to discard 51% of 10,750 species trends derived from occupancy modelling due to insufficient recording coverage. There are social reasons for increasing citizen science participation too, as it can yield benefits for the biodiversity conservation movement and wider society (Bela *et al*. 2016; Coventry *et al*. 2019). But there can be trade-offs (e.g. in the quality of sampling and data) as well as benefits to the participation of citizen scientists in biodiversity monitoring. In this thesis I explore some of these pros and cons, both of which could be magnified by increased participation. First, I review evidence from the literature for moth biodiversity trends and the potential drivers of these trends, highlighting knowledge gaps that could be addressed by increased citizen science recording (Chapters 2 and 3). Next, I consider the reliability of mass-participation citizen science engaging inexperienced contributors in comparison to standardised monitoring by expert amateur naturalists, by comparing population changes for widespread butterfly species from the Big Butterfly Count with those from the UKBMS (Chapter 4). Finally, I examine the implications of involving citizen scientists in gathering ecological information for the reuse of data (Chapter 6).

Citizen science has played an important role historically in understanding biodiversity change and has the potential to be even more significant (Powney & Isaac 2015; Pocock *et al*. 2018). I present several new applications of citizen science data using GB butterflies and moths as examples. Utilising the NMRS, I produce the first long-term occurrence trends for macro-moths (Chapter 3) and make inferences about the drivers of change. Second, I use the Big Butterfly Count data to estimate trends for a group of widespread butterfly species (Chapter 4). Then, I use UKBMS and RIS monitoring data to explore the variability of short-term population trends and how this impacts on Red List classifications (Chapter 5). Lastly, I investigate how the opinions of citizen scientists may limit the wider use of Lepidoptera records (Chapter 6).

## Aims, results and contribution to scientific knowledge of each chapter

### *Chapter 2 The decline of moths in Great Britain: a review of possible causes*

Moths are a species-rich insect taxon (with c.2,500 species recorded in GB) and play important roles in ecosystem functioning, as herbivores (Young 1997; Majerus 2002), prey for a wide range of predators (Vaughan 1997; Denerley *et al*. 2019; Rytkönen *et al*. 2019) and as pollinators (Banza *et al*. 2015; Macgregor *et al*. 2015). Previous research has shown significant decreases in GB moth abundance (Conrad *et al*. 2006; Fox *et al*. 2013) and comparable studies elsewhere have produced similar evidence of overall decline (Mattila *et al*. 2006; Groenendijk & Ellis 2011; Valtonen *et al*. 2017).

Understanding the causes of moth decline and the potential impacts on other species is of high policy relevance (Sutherland *et al*. 2006) and current knowledge gap (Burns *et al*. 2016). I summarise moth biodiversity trends, review the evidence for the potential drivers of these trends and recommend future research.

Although few scientifically documented examples exist, 20th century land-use change, converting semi-natural habitats to highly-modified landscapes, will have had major detrimental effects on many moth species. Even such overwhelmingly negative (for biodiversity) land-use changes have provided opportunities for some moths however, e.g. species that utilise conifers as larval hostplants. More evidence exists for adverse impacts of changes to the intensity of land management, particularly agricultural intensification and a shift from traditional silviculture techniques such as coppicing to high forest systems. In contrast, apparent climate change effects on moths are largely positive, reflecting the fact that many species reach their cool (northern) range margin within GB. At the time of the review, there was little evidence of population-level effects on moths caused by chemical or light pollution, non-native species or direct exploitation.

I recommend more research into the impacts of light pollution, climate change and trying to disaggregate the effects of different elements of intensive land management. A broader view of GB moth trends is also recommended, beyond the selection of widespread species for which RIS trends are available, and evidence-based habitat improvement measures need to be developed and implemented via agri-environment schemes and other policy initiatives.

Although previous studies reporting moth trends had given brief overviews of suspected causes, the review presented in Chapter 2 provided a comprehensive statement of current knowledge. Much progress has been made since it was published (summarised in section 1.6), particularly into the effects of climate change, artificial light and, to a lesser extent, nitrogen pollution. A broader assessment of GB moth trends has been achieved, first through the analysis presented in Chapter 3 and more recently in Randle *et al*. (2019).

### *Chapter 3 Long-term changes to the frequency of occurrence of British moths are consistent with opposing and synergistic effects of climate and land-use changes*

As identified in Chapter 2, it is important to assess trends of GB moths because of the taxon’s species richness and significant ecological roles. Such trends identify priorities for biodiversity conservation. In addition, the patterns of change can provide insight into the causal factors. Latitudinal gradients of two major drivers implicated in moth trends exist within GB; a natural climatic gradient, with warmer temperatures in the south compared with the north, and a land-use intensity gradient, with generally higher levels of intensive agriculture and urban development in the south compared with the north. By examining moth trends across these gradients, some inferences can be drawn about the contributions of drivers.

Utilising 10.5 million NMRS distribution records, I estimate GB trends for 673 resident macro-moth species for 1970-2010. The Frescalo statistical approach was used to account for major spatiotemporal variation in recording effort (Hill 2012). Overall, moths decreased in frequency of occurrence, but individual species exhibited a wide diversity of responses with 260 species showing significant declines and 160 significant increases. Northerly distributed (cold-adapted) species declined, consistent with a negative response to climate change in landscapes relatively unaffected by intensive land management, while (warm-adapted) moths restricted to southern GB showed more mixed results, consistent with expected distribution increases driven by climate change but detrimental impacts of more intensive land-use. Widespread species, which were not predicted to be as sensitive to climatic change, declined on average in southern GB but not in the north, suggesting a response to land-use change. A traits-based analysis (using Ellenberg indicator values) of the larval hostplants of monophagous moth species, found a significant positive correlation between moth distribution trend and the nitrogen requirements of hostplants and a negative association with light requirements.

This study considerably extends previous knowledge of GB moth biodiversity change by estimating long-term trends for 673 species, rather than 337 species with RIS abundance trends (Conrad *et al*. 2006; Fox *et al*. 2013). This improvement is more than simply numerical, because the new assessment includes scarcer species, which may be more likely to be at risk, than those in the RIS analysis and by including diurnal moths and species that do not exhibit positive phototaxis. The study reveals likely positive and negative responses to climate change by warm- and cold-adapted moths (respectively), mirroring those of butterflies (Hill *et al*. 2002; Franco *et al*. 2006) and opposing effects of climatic and land-use change in southern GB. It also provides one of the first demonstrations of nitrogen enrichment impacts on Lepidoptera populations mediated via plant communities (Öckinger *et al*. 2006; de Sassi *et al*. 2012; Betzholtz *et al*. 2013).

### *Chapter 4 Using citizen science butterfly counts to predict species population trends*

Citizen science is a nebulous discipline (Pocock *et al*. 2017), even in the limited context of biodiversity monitoring. The long-standing collation of species population and distribution data in GB falls under a broad definition of citizen science, as most data are gathered by unpaid volunteers. However, these volunteers are often skilled and experienced, capable of undertaking monitoring to comparable standards as professional scientists (Chase & Levine 2016) and the results are generally accepted by policy makers and the scientific community. In contrast, in the recent proliferation of citizen science projects, many participants have little prior experience; indeed some biodiversity projects explicitly aim to engage with new audiences (e.g. Roy *et al*. 2016).

As a consequence of inexperienced participants and simplified sampling protocols, the reliability of data gathered through mass-participation citizen science may be constrained and the credibility of scientific outputs called into question (Gardiner *et al*. 2012; Riesch & Potter 2014; Lewandowski & Specht 2015; Kosmala *et al*. 2016). There have been few attempts to compare population trends generated by mass-participation citizen science against those from systematic monitoring, and none involving terrestrial invertebrates.

I address this knowledge gap by determining whether population changes for widespread butterfly species derived from a new mass-participation scheme, the Big Butterfly Count, are comparable with those from an established, highly structured programme based on expert data, the UKBMS. Using data just for the three-week Big Butterfly Count survey period each year, I found significant correlations between Big Butterfly Count and UKBMS trends for both net population change over a four-year period (2011-2014) and inter-annual population growth rates. Furthermore, linear models using Big Butterfly Count data and weather covariates were surprisingly successful at predicting UKBMS index values in 2015.

The findings inform the debate around the ability of mass-engagement projects to produce robust scientific outputs in addition to improving scientific literacy, awareness and engagement (Chase & Levine 2016; Lakeman-Fraser *et al*. 2016; Turrini *et al*. 2018). There is also much interest in the feasibility of using simple, non-systematic sampling to generate meaningful biodiversity information (e.g. Lang *et al*. 2019). In addition, the validation of Big Butterfly Count trends means that the data could be used as a separate indicator of GB butterfly populations or incorporated into existing UKBMS metrics (e.g. Dennis *et al*. 2017c) with an integrated analysis (Pagel *et al*. 2014). The Big Butterfly Count samples different landscape elements than the UKBMS; the majority of Big Butterfly Counts are undertaken in private gardens (a land-use type not sampled by the UKBMS) and occur, on average, in more urban settings. Thus, Big Butterfly Count data could potentially be used as an indicator of butterfly populations in gardens and parks. This could provide a valuable tool to engage the public and managers of urban greenspace to encourage more favourable land management for biodiversity (Garbuzov *et al*. 2015; Gunnarsson *et al*. 2017), as well as contributing to increased understanding of urban ecology (Wang Wei *et al*. 2016) and the importance of built-up areas for insects in highly modified landscapes (Baldock *et al*. 2015; Hall *et al*. 2017).

### *Chapter 5 Insect population trends and the IUCN Red List process*

The International Union for Conservation of Nature’s (IUCN) Red List process plays a vital role in biodiversity assessment, as a set of objective standards for quantifying extinction risk (Mace *et al*. 2008), assessing change (Butchart *et al*. 2005) and catalysing conservation (Rodrigues *et al*. 2006). Despite its undoubted utility and widespread application, the Red List process is not without problems. Some stem from misunderstanding (Collen *et al*. 2016), while others arise from debate over the quantitative thresholds of certain Red List criteria. Criterion A “Reduction in population size”, for example, determines extinction risk solely on the basis of population decline over the most recent 10 years or three generations, whichever is longer (IUCN 2012). Several studies have questioned the reliability of measuring trends over such a short period

(Connors *et al*. 2014; d'Eon-Eggertson *et al*. 2015; White 2019) and cautioned that long time series may be necessary to detect impacts of drivers and changes in ecosystem functioning (McCain *et al*. 2016; Thomson 2019). A recent study of GB moth biomass concluded that short durations of data generate unreliable estimates of longer-term trends (Macgregor 2019a).

Given that Lepidoptera typically have high population variability (Williams 1961; Taylor & Taylor 1977), I hypothesised that 10-year population trends of butterflies and moths, and resultant IUCN classifications, would be sensitive to start year. I explore this using UKBMS and RIS citizen-science data to derive a series of 10-year trends with different start years for 54 butterfly and 431 macro-moth species. Each trend was then compared to IUCN Criterion A thresholds to produce Red List classifications for each date period. Large discrepancies were revealed between classifications that differed by just a single start year. For example, 15 butterfly species met the Red List threshold using trends for 2002-2011, but 29 did so for 2003-2012. In the most extreme example, the difference of a single year reduced the number of qualifying moth species from 62 to 20.

Current IUCN guidelines acknowledge that using data from a longer time period may be advantageous for species that have high population variability (IUCN 2017). However, 10-year trends remain the basis of Criterion A and can be applied without utilising longer-term data. Previous authors have noted this potential problem (de Iongh & Bal 2007; van Swaay *et al*. 2011), but this is the first time that such impacts have been quantified for insect taxa. My results suggest that it is inappropriate to use 10-year trends in extinction risk assessment of UK Lepidoptera and that this is likely to be the case for many other insect taxa globally. I call for further guidance on Red List assessments of taxa with high levels of population variability.

*Chapter 6 Opinions of citizen scientists on open access to UK butterfly and moth occurrence data*

Recent studies have examined the motivations of citizen scientists (Hobbs & White 2012; West & Pateman 2016; Richter *et al*. 2018), the benefits that they gain from participation (Merenlender *et al*. 2016; Coventry *et al*. 2019) and impacts on their engagement with conservation (Lewandowski & Oberhauser

2017). However, little work has been undertaken on the views of citizen scientists regarding the onward use of the data they contribute, beyond the project in which they participated (Ganzevoort *et al*. 2017). This is topical due to scientific and ethical pressure for data sharing (Hampton *et al*. 2013), and requirements imposed by public funding bodies and scientific journals for open access to datasets (Reichman *et al*. 2011; Pearce-Higgins *et al*. 2018). It is often suggested, without supporting evidence, that citizen scientists favour open access, but at present many citizen science biodiversity datasets are not fully open (Groom *et al*. 2017).

Through questionnaire surveys of two groups of citizen scientists involved in the BNM and NMRS projects, namely recorders (510 survey respondents) and regional co-ordinators (County Recorders) (104 survey respondents), I characterise views relating to open access. Overall, I found high levels of support for the principle of open access to UK butterfly and moth records; e.g. more than twice as many regional co-ordinators (39.8%) were classified as promoters of open access compared with detractors (16.5%). Despite this, there was much more caution when it came to the practicalities. Only 6.7% of regional co-ordinators and 32.7% of recorders thought that all records should be open at full spatial resolution, and 79.6% of regional co-ordinators felt that data reuse should be limited to non-commercial purposes. There were significant regional differences, however, with co-ordinators in Scotland being more supportive of open access than their counterparts in England.

Knowledge of these opinions, including details about which data should be accessible, when and for what purposes, contributes to the current debate on open access taking place within organisations collecting, curating and utilising species records provided by citizen scientists. Open access to such data would maximise their use in biodiversity conservation (Chandler *et al*. 2017; Sullivan *et al*. 2017; Soroye *et al*. 2018), contribute to ecological research (Farley *et al*. 2018) and increase public trust in science (Soranno *et al*. 2015). Unlike in conventional science though, where researchers are also data gatherers and are able to decide on issues of data access, in citizen science, researchers should also be mindful of the views of the participants. There can be negative consequences of open access to biological records, e.g. harm to threatened species and habitats (Tulloch *et al*. 2018), and citizen scientists might take the view that the risks outweigh the benefits and cease participation, undermining project viability. My results, and those from the Netherlands (Ganzevoort *et al*. 2017), suggest that participants expect some limitation on data availability. Organisers of citizen science should consider open access issues in the project planning phase and present would-be participants with clear information about onward data availability from the outset.

### Future directions

Although much has been achieved using citizen science data to assess the magnitude and causes of biodiversity change in the case study of GB butterflies and moths, there remains huge potential for further improvement in generating reliable trends and, particularly, in extending geographical and taxonomic scope. Citizen science provides cost-effective biodiversity surveillance (Gardiner *et al*. 2012) but necessitates careful consideration of data quality and specialised analysis to adjust for bias. Data are currently inadequate to assess the vast majority of species in most countries. GB Lepidoptera are intensively recorded by citizen scientists and yet major gaps in our knowledge remain. Long-term population trends are restricted to c.400 of the c.2,500 GB moth species, and no distribution trends exist for the c.1,600 micro-moths. Increased recording and novel statistical methods will both contribute to the production of robust trends for more GB moths in the future. The extensive datasets on GB Lepidoptera provide a model system for the development of new statistical approaches (e.g. Dennis 2017a) that can be applied to other taxa, and for comparing different methods (Norberg *et al*. 2019). Further research is needed to generate meaningful short-term trends for taxa with highly variable population growth rates. We also need to explore the extent to which mass-participation citizen science can inform species trends and, indeed, whether it can provide additional information about biodiversity change in poorly-monitored habitats such as gardens.

Extending citizen science approaches to other taxa and wider geographical scales is an important next step towards more informative and dependable assessments of global insect biodiversity change (Montgomery *et al*. 2020; Harvey *et al*. 2020). In Europe, there are currently several significant developments, including the ABLE project (https://butterfly-monitoring.net/able) to extend standardised butterfly transect monitoring to new countries and to roll-out a mass-participation butterfly recording scheme, collating data into a pan-European database (https://butterfly-monitoring.net/), and the European Ladybird Survey (https://european-ladybirds.brc.ac.uk/home) with a new smartphone app to facilitate citizen science recording. However, the tropics represent the key information gap and areas such as southern Asia and tropical Africa have been identified as regions where increased citizen science could provide the greatest benefits for global biodiversity assessment (Pocock *et al*. 2018). Scaling-up citizen science to address this gap is a crucial challenge over the next decade (Chandler *et al*. 2017).

New technologies such as eDNA (Ruppert *et al*. 2019) and image recognition cameras (Hogeweg *et al*. 2019) may revolutionise aspects of biodiversity monitoring, but the direct engagement of citizens in gathering data will remain vital to counteract the “extinction of experience” (Miller 2005; Soga & Gaston 2016) and “shifting baseline syndrome” (Soga & Gaston 2018). Citizen science can engender support for biodiversity conservation and encourage civic participation (Turrini *et al*. 2018), although these outcomes need to be measured more effectively (Bela *et al*. 2016). Co-created projects (Trimble & Berkes 2013; Pocock *et al*. 2018), citizen science within formal education (Wals *et al*. 2014; Saunders *et al*. 2018) and balanced reporting of results (McAfee *et al*. 2019) will all build trust in biodiversity science and help deliver the transformative change required to tackle current environmental crises (Díaz *et al*. 2019).

Finally, more needs to be done to make citizen science biodiversity data available to support wider research, policy development and implementation (Chandler *et al*. 2017; Sullivan *et al*. 2017; Montgomery *et al*. 2020). Research priorities should include identifying the relative contributions of anthropogenic drivers to insect declines (which are incompletely understood even for many GB Lepidoptera; Chapter 2) and the design of evidence-based land management techniques to restore biodiversity, even if drivers are not fully known (Harvey *et al*. 2020). There are barriers to overcome in increasing access to citizen science data and new initiatives need to address these from the outset, while long-running projects, such as those for GB butterflies and moths, should work with participants to promote the benefits of open data and mitigate any negative impacts (Tulloch *et al*. 2018). Immediate actions could include focus groups with GB Lepidoptera recorders to clarify their views on open data e.g. participants favoured only permitting non-commercial reuse of data, but it is unclear what they regard as commercial use of biodiversity information.

## Conclusions

Global biodiversity is in steep decline, although data are currently insufficient to estimate rates of change reliably for insects, leading to some injudicious predictions of imminent ‘insect armageddon’. In GB however, butterflies and moths are among the best monitored taxa and show clear overall decreases since the 1970s, albeit with a minority of species faring well. Citizen science already makes an enormous contribution to knowledge of biodiversity change in some countries and has the potential to provide much-needed data from poorly-studied regions in order to give a more representative global perspective.

Building on the strong tradition of citizen science Lepidoptera recording in GB, I show that such data can be used to produce a comprehensive assessment of distribution trends for macro-moths (Chapter 3) and robust measures of population change for widespread butterflies (Chapter 4). The former provides essential information to input into the prioritization of moth species for conservation action, vital given the overall declines of moths in GB, and also sheds light on drivers, such as land-use change, climate change and nutrient enrichment, helping to fill knowledge gaps (Chapter 2). Through these analyses, I show that major increases in citizen science engagement can be achieved, benefitting participants and increasing support for biodiversity conservation, without compromising scientific outputs.

The involvement of citizen scientists enables biodiversity monitoring at large spatial and temporal scales. Long-term monitoring is particularly important in the assessment of abundance trends for species with high levels of population variability (Chapter 5); short-term trends, even those measured over the Red List 10-year standard, are easily skewed by frequent peaks and troughs in abundance. However, the involvement of citizen scientists may also place novel constraints on researchers, unless these are circumvented by project design. I explore one such issue, showing that citizen scientists expect restrictions on the reuse of their records, limiting the benefits of open data (Chapter 6).

Biodiversity loss is a crisis with drastic implications for human society. Citizen science, by providing scientific data to demonstrate and monitor biodiversity loss and by engaging citizens directly with the issue, can play a significant role in attempts to avoid the worst of the predicted impacts and bring about transformative change for a brighter future.

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## Chapter 2: The decline of moths in Great Britain: a review of possible causes

Slightly modified from:
Fox R (2013) The decline of moths in Great Britain: a review of possible causes. *Insect Conservation and Diversity* **6**, 5–19.

### Abstract

Population declines among insects are inadequately quantified, yet of vital importance to national and global biodiversity assessments and have significant implications for ecosystem services. Substantial declines in abundance and distribution have been reported recently within a species-rich insect taxon, macro-moths, in Great Britain and other European countries. These declines are of concern because moths are important primary consumers and prey items for a wide range of other taxa, as well as contributing to ecosystem services such as pollination.

I summarise these declines and review potential drivers of change. Direct evidence for causes of moth declines is extremely limited, but correlative studies and extrapolation from closely related taxa suggest that habitat degradation (particularly because of agricultural intensification and changing silviculture) and climate change are likely to be major drivers. There is currently little evidence of negative population-level effects on moths caused by chemical or light pollution, non-native species or direct exploitation.

I make suggestions for future research with a focus on quantifying impacts of land management practices, light pollution and climate change on moth population dynamics and developing evidence-based measures that can be incorporated into agri-environment schemes and other policy initiatives to help reverse the widespread decline of moths in Great Britain and beyond.

### Introduction

The Earth is undergoing a period of substantial decreases in biodiversity and mass extinction of species (Pimm *et al*. 1995; Dirzo & Raven 2003; Butchart *et al*. 2010; May 2010; Mooney 2010), which threaten ecosystem services and the welfare of the human race (Balmford & Bond 2005; Millennium Ecosystem Assessment 2005; Schröter *et al*. 2005; Biesmeijer *et al*. 2006; Rockström *et al*. 2009; UK National Ecosystem Assessment 2011). However, the decline and extinction rates of insects, which comprise the majority of terrestrial biodiversity, are inadequately quantified and poorly understood (McKinney 1999; Dunn 2005; Thomas 2005). A contributory factor to this knowledge gap is the highly variable population dynamics of many insect species (Wilson & Roy 2009). Long time series of data are required to identify significant directional trends amid the statistical ‘noise’ of population cycles and short-term responses to stochastic environmental events (Conrad *et al*. 2004).

Until recently, large spatial-scale assessments of long-term insect trends were restricted to a few charismatic, well-studied, but species-poor, taxa such as butterflies and bumblebees, in some developed nations, particularly in western Europe (Maes & Van Dyck 2001; Warren *et al*. 2001; Fox *et al*. 2006a; Fitzpatrick *et al*. 2007; Kosior *et al*. 2007; Goulson *et al*. 2008; van Swaay *et al*. 2008; Van Dyck *et al*. 2009; Cameron *et al*. 2011). Thomas *et al*. (2004b) showed that butterfly declines exceeded comparable changes among birds and vascular plants in Great Britain (GB). These examples provide insight into insect diversity trends, but concerns remain over how representative they are across insect taxa (e.g. Hambler & Speight 2004; but see Thomas & Clarke 2004).

Recently, studies of moths have generated the first evidence of national-scale declines in a species-rich insect taxon (Conrad *et al*. 2006; Mattila *et al*. 2006, 2008; Groenendijk & Ellis 2011). Such studies are important as they corroborate the use of taxa such as butterflies as indicators of wider insect biodiversity trends, but also because the greater diversity of moths may facilitate an improved understanding of the drivers of change and the impacts that insect declines will have on other organisms, communities and ecosystem functioning.

This article reviews current knowledge about moth declines and the potential drivers of change in GB (and elsewhere in western Europe). The five main causes of biodiversity loss and changes in ecosystem services in the UK (UK National Ecosystem Assessment 2011) are examined in relation to moths, as well as an additional potential driver, light pollution. This review is topical at a time of resurgent interest in ecological research on moths, stimulated, in part, by the discovery of the widespread and substantial declines in this taxon (Sutherland *et al*. 2006).

### Moth declines in GB and beyond

Although national-level extinctions (Parsons 2003) and decreased distribution and abundance of selected diurnal macro-moths had already been documented (Groenendijk & van der Meulen 2004), the analyses of the Rothamsted Insect Survey (RIS) monitoring data yielded the first quantitative understanding of the severity of population decline among moths.

The RIS, a nationwide network monitoring UK moth populations, has been operated by Rothamsted Research since 1968 and provides one of the longest-running and most spatially extensive datasets of a species-rich insect taxon anywhere in the world (Conrad *et al*. 2007; Woiwod & Gould 2008). Monitoring at one RIS site (Rothamsted, UK) commenced in 1933 and has demonstrated a substantial decrease in abundance and diversity of moths during the 1950s (Woiwod & Gould 2008). Furthermore, detailed national studies of an individual species, Garden Tiger *Arctia caja*, also demonstrated severe population and site occupancy declines for this once-common species and paved the way for a more comprehensive assessment (Conrad *et al*. 2002). Analysis of a 35-year dataset (1968-2002) for 337 macro-moth species (those for which adequate data were available) revealed significant decreases (Conrad *et al*. 2004). The total abundance of individual macro-moths caught by the RIS network decreased in the whole of GB (31% decrease over 35 years) and in southern GB (44% decrease) (Conrad *et al*. 2006). The total abundance of moths did not decrease in northern GB, a finding corroborated by a smaller study of RIS data from a single site by Salama *et al*. (2007) and also by butterfly trends (Brereton *et al*. 2011).

Conrad *et al*. (2006) also found that 66% of the 337 species studied had negative population trends and that 21% of the species had decline rates >30% 10 year$^{-1}$ (equivalent to the IUCN threshold levels for Red List threat categories). That these 337 species are considered widespread and generally common in GB (Skinner 2009; Waring *et al*. 2009) underscored the significance of Conrad *et al*.’s findings for biodiversity conservation (Fox *et al*. 2006b). A similar proportion of species had undergone substantial population decreases in northern GB and southern GB, but many more species had increased in the north and this appears to account for the lack of a significant trend in overall moth abundance there compared to the south (Fox *et al*. 2006b).

Parallel decreases in the abundance or distribution of macro-moths have now been reported from other European countries. Groenendijk and Ellis (2011) found a pattern of change among 733 macro-moth species in the Netherlands, which was strikingly similar to the British findings: 71% of Dutch species decreased in abundance and the total abundance of moths decreased by one-third (1980-2009). Both studies also highlighted a minority of species that had markedly increased in abundance.

Utilising long-term distribution records, Mattila *et al*. (2006, 2008) showed significant overall decreases in the distribution of macro-moths in the families Geometridae and Noctuidae (590 species in total) in Finland and a study of Lepidoptera at a nature reserve in southern Sweden revealed high rates of local extinction over a 50-year period (27% of 597 study species were deemed to have become extinct versus 4% that had colonised the area) (Franzén & Johannesson 2007). Preliminary analysis of the new National Moth Recording Scheme dataset in the UK also indicated severe distribution declines among some macro-moth species (Fox *et al*. 2011b).

Several of these studies examined ecological traits and life history attributes in relation to rates of distribution or population change, but the results varied considerably. For example, in GB and the Netherlands, species overwintering in the adult life-cycle stage had positive population trends over time (Conrad *et al*. 2004; Groenendijk & Ellis 2011), whereas Mattila *et al*. (2006) found adult overwintering to be a significant predictor of increased extinction risk and Franzén and Johannesson (2007) found no effects of overwintering strategy on species persistence. However, range size and larval specificity correlated consistently with rates of decline or extinction risk, mirroring studies on butterflies (Warren *et al*. 2001; Koh *et al*. 2004; Nilsson *et al*. 2008). Rarer species were associated with greater losses or increased likelihood of extinction

(Franzén & Johannesson 2007; Groenendijk & Ellis 2011) and monophagous species were more likely to have declined or become extinct than less-specialised species (Franzén & Johannesson 2007; Mattila *et al*. 2008).

Taken together, these studies provide overwhelming evidence of moth declines on a large geographical scale and mirror previous studies of less species-rich taxonomic groups such as butterflies. Such losses are likely to have substantial impacts at higher and lower trophic levels, because of the importance of moths as herbivores, pollinators and prey items (e.g. Proctor *et al*. 1996; Vaughan 1997; Wilson *et al*. 1999; Wickramasinghe *et al*. 2004; Devoto *et al*. 2011) and may affect the delivery of some ecosystem services. Yet, the causes of pervasive moth declines are poorly understood.

## Drivers of change in moth populations

### *Habitat loss, degradation and fragmentation*

The destruction and modification of habitats by human activity is regarded as the foremost cause of global biodiversity loss (Diamond *et al*. 1989; Brooks *et al*. 2002; Dirzo & Raven 2003; Fahrig 2003). Habitat loss (including deterioration in quality and the isolation effects of fragmentation) has also been identified as the principle driver of butterfly declines in Europe (e.g. Asher *et al*. 2001; Maes & Van Dyck 2001; Warren *et al*. 2001; Wenzel *et al*. 2006; Bulman *et al*. 2007; Hanski & Pöyry 2007; Van Dyck *et al*. 2009; Öckinger *et al*. 2010). Consequently, it seems probable that habitat loss will have influenced moth abundance and distributions in GB (Fox *et al*. 2006b), although habitat degradation patterns vary geographically and, therefore, impacts on species are expected to differ between areas. It is possible that the better performance, on average, of moth populations in northern GB stems from lower levels of habitat degradation relative to the southern half of GB, although a climatic explanation, or a combination of both, is also plausible (see section Climate change and Chapter 3).

There is little direct evidence for habitat loss, degradation or fragmentation effects on moth populations in GB (or elsewhere). However, as for butterflies, there is considerable circumstantial evidence that the widespread destruction of semi-natural habitats has had a severe impact on specialist moths, and it has been implicated in the extinction of species, including Reed Tussock *Laelia coenosa* and Gypsy Moth *Lymantria dispar* because of wetland drainage, and Spotted Sulphur *Acontia trabealis* as a result of afforestation and agricultural intensification (Majerus 2002). Habitat changes may also have played a role in the declines of species such as *Pyrausta sanguinalis* in sand dunes, Straw Belle *Aspitates gilvaria* and Black-veined Moth *Siona lineata* on unimproved grassland, Shoulder-striped Clover *Heliothis maritima* and Speckled Footman *Coscinia cribraria* on lowland heath and Barberry Carpet *Pareulype berberata* in hedgerows (Fox *et al*. 2010).

Fragmentation effects have been detected in few empirical studies of moths (Öckinger *et al*. 2010), but generally biodiversity impacts from fragmentation *per se* tend to be relatively small compared to the effects of habitat loss and habitat quality (Thomas *et al*. 2001; Fahrig 2003; Hodgson *et al*. 2009). In addition, theory predicts that mobile species are less likely to experience negative effects of isolation. Mobility is poorly understood in most moth species (apart from long-distance migrants, e.g. Chapman *et al*. 2011), but recent evidence suggests that many species are relatively mobile (Franzén & Nilsson 2007; Merckx *et al*. 2009a, 2010a,b; Betzholtz & Franzén 2011; Slade *et al*. 2013; but see Nieminen 1996; Nieminen *et al*. 1999). Thus, while fragmentation might be expected to be important for some specialised species with low to intermediate mobility (Thomas 2000), it is unlikely to be a principle driver of the declines of many widespread moths in GB and elsewhere.

In contrast, it seems highly plausible that the widespread destruction of semi-natural habitats that took place across GB during the 20th century had substantial impacts on moths. These were rarely documented through site-based population monitoring at the time (although see Woiwod & Gould 2008), and land-use change effects cannot easily be assessed retrospectively. However, recent research has started to shed light on the impacts of land use on moth populations, by contrasting different levels of management intensity.

*Agricultural management*. Agriculture is a dominant and socioeconomically important land use in GB and much of Europe and is also of great importance for biodiversity (Bignal & McCracken 1996; Halada *et al*. 2011). However, agricultural intensification generally reduces habitat area, quality and heterogeneity through the interlinked impacts of increased agrochemical use, changes in tillage/grazing practices and larger cropped areas and is widely recognised as a major driver of biodiversity decline (Donald *et al*. 2001; Benton *et al*. 2002, 2003; Robinson & Sutherland 2002; Kleijn *et al*. 2009). The substantial drop in moth abundance and diversity recorded on farmland at Rothamsted between the 1940s and 1960s was concomitant with agricultural intensification of the surrounding land (Woiwod & Gould 2008). Specific changes included a move from grassland to arable cultivation, removal of hedgerows and uncultivated areas to increase field size and built development. A number of other recent studies have also implicated aspects of intensification with reduced moth populations (see below).

Taylor and Morecroft (2009) reported significant increases in moth abundance and species richness on a farm in southern England, following organic conversion and simultaneous entry into an agri-environment scheme (AES) and the adoption of less-intensive farming techniques. Wickramasinghe *et al*. (2004) found significantly higher species richness and diversity of moths on organic farms than on conventional ones in a study of 24 pairs of (livestock and mixed) farms in GB. The authors ascribed this difference to the reduced use of agrochemicals, but many other factors could also be responsible. Pocock and Jennings (2008) conducted a similar study, but were able to separate out several different elements of intensification. They found the greatest effects on moth abundance related to the presence or absence of field boundaries (moths benefited from boundaries), both in arable and in pasture fields, with relatively little impact from either agrochemical inputs or the switch from hay to silage cropping regimes. This corroborates findings that the area of hedges and bushes in the local environment around RIS traps on the Rothamsted Estate was an important predictor of moth abundance and diversity (Woiwod & Gould 2008).

Work by Merckx *et al*. (2009a,b, 2010a,b) also highlighted the importance of field boundaries for moths in agricultural settings. The presence of hedgerow trees and 6m-wide grassy field margins were both significantly correlated with increased moth abundance and diversity (Merckx *et al*. 2009b). Such field margins, but not hedgerow trees, were management options for which ‘entry level’ AES payments were available at the time of the studies. Hedgerow trees had the greater effect, but only when targeted management advice resulted in elevated levels of AES uptake in the surrounding landscape (Merckx *et al*. 2009b). Hedgerow trees had a positive impact on a wide range of moths, not just those species that utilise them as larval hostplants, possibly because they provide sheltered micro-climates in relatively exposed landscapes (Merckx *et al*. 2010a).

Another study (Fuentes-Montemayor *et al*. 2011) found benefits for moths from AES management at farms in Scotland. Conversion of conventional arable or improved pasture fields to more species-rich grassland under AES resulted in increased abundance and species richness of moths. Other AES options, including the creation of extensively managed margins, also led to increased moth numbers and abundance, but no effects were found for AES hedgerow management.

Agricultural use of chemicals, both fertilizers and pesticides, increased enormously as an integral part of agricultural intensification during the latter half of the 20th century. With direct and indirect (e.g. via impacts on larval hostplants, nectar sources, vegetation structure and composition) effects on many taxa both within cropped areas and on field margins (Freemark & Boutin 1995; McLaughlin & Mineau 1995; Longley & Sotherton 1997), these agrochemicals may have played a prominent role in the decline of moths in GB. However, disentangling the relative contributions of fertilizers or pesticides from other elements of agricultural intensification at a landscape or national scale is problematic (Benton *et al*. 2003; although see Gibbs *et al*. 2009).

Ongoing agricultural development will alter patterns of agrochemical use and the nature of the substances deployed. Such changes may increase or decrease potential impacts on biodiversity and should be evaluated prior to introduction. For example, genetically modified herbicide-tolerant crops alter pesticide regimes and aim to improve the efficacy of weed control, with potential impacts on plants and associated invertebrates both within the crop and on field margins (Roy *et al*. 2003). Novel crops (e.g. biofuel and biomass), increasing resistance to pesticides and changing food security conditions may drive increased intensification and additional exposure to existing and future agrochemicals (Sutherland *et al*. 2008).

Often, subtle aspects of habitat quality are vital for population persistence. Change in the grazing intensity of agricultural land is known to alter habitat quality critically for many taxa, including butterflies, vascular plants and some specialist moth species. For example, increased intensity of livestock grazing almost led to the extinction of New Forest Burnet *Zygaena viciae* from GB (Young & Barbour 2004). Experimental reduction of the high intensity of livestock grazing typical of commercial upland agriculture led to significant increases in moth abundance and species richness (Littlewood 2008). While less-intensive grazing may benefit grassland insects, the permanent abandonment of traditional pastoral agriculture, leading to rapid ecological succession, can be detrimental (Balmer & Erhardt 2000; Bourn & Thomas 2002; Öckinger *et al*. 2006; van Swaay *et al*. 2006; Settele *et al*. 2009; Stefanescu *et al*. 2009). Such abandonment is thought to have contributed to declines of moth species in GB such as Forester *Adscita statices* and Narrow-bordered Bee Hawk-moth *Hemaris tityus* (M. Parsons pers. comm.).

*Woodland management*. Native broad-leaved and coniferous woodlands are important habitats for a wide range of taxa in GB, including a high proportion of the macro-moth species. Although woodlands of high biodiversity value have been destroyed, the net amount of broad-leaved woodland has increased in GB over recent decades, in stark contrast to the amount of other semi-natural habitats. And yet, the changing status of key monitored taxa, such as birds, butterflies and plants, clearly indicates a decrease in woodland biodiversity (Fuller *et al*. 2005; van Swaay *et al*. 2006; Carey *et al*. 2008; Fox *et al*. 2011a). A range of factors are responsible for these declines but, for butterflies, the main causes appear to be altered structural diversity, botanical communities and micro-climatic conditions associated with a shift towards high-forest management (including the cessation of traditional practices such as coppicing), leading to increasing shade and fewer open, early-successional habitats (Warren & Key 1991; Sparks *et al*. 1996; Asher *et al*. 2001; van Swaay *et al*. 2006; Clarke *et al*. 2011). Conrad *et al*. (2004) found that moth species utilising deciduous trees as larval hostplants tended to have negative population trends in GB, while the few species (such as Spruce Carpet *Thera britannica* and Pine Beauty *Panolis flammea*) that exploit coniferous trees generally increased. The latter is hardly surprising, given the massive expansion of conifer plantations (a 20-fold increase, 1800-1980) in GB.

Moth species assemblages vary between woodland types and along geographical gradients, but also within woods (e.g. species associated with mature trees, others with edge habitats or open, grassland conditions in rides and glades) and even between age-classes of managed areas such as coppice coupes (Broome *et al*. 2011).

Merckx *et al*. (2012) assessed the macro-moth response to standard woodland conservation management practises in a landscape-scale study in southern England. They found that moth abundance increased with the amount of shelter: open, recently coppiced areas had the lowest abundance and standard (narrow) forest rides and blocks of mature woodland had the highest. However, common management techniques to open up woodland for the benefit of taxa such as butterflies, including coppicing and ride widening, did benefit the overall species richness of moths in the woodland landscape. Wide rides, although containing relatively low abundance levels of moths, were as rich in species as the standard rides and mature woodland. Moreover, the introduction of increased structural and micro-climatic heterogeneity increased overall species richness by providing niches for moths that were not found elsewhere in the woods. The authors caution, however, against opening up the sheltered late-successional cores of woodlands as these support high abundance and species richness of many specialist and conservation priority moths that are not found in more open habitats.

Most woodland specialist moths may have benefited from the switch to high-forest management in broad-leaved woodland habitats over recent decades, although they will have been impacted detrimentally by conversion to coniferous forestry. However, it is equally clear that many moths, mostly generalist species of more open habitats (but also some specialists such as *Anania funebris* and

Drab Looper *Minoa murinata*) will have undergone substantial decreases in abundance and distribution as a result of changing woodland management.

*Urbanisation*. The impacts of urbanisation on biodiversity are complex. Increasing urban land cover typically replaces and fragments semi-natural habitat, leading to decreases in biodiversity, particularly among specialist species (Bergerot *et al*. 2010; Gaston & Evans 2010; UK National Ecosystem Assessment 2011). However, urbanisation can also cause increases in biodiversity among particular taxa (McKinney 2008). In addition to habitat loss, urbanisation also generates other environmental changes that might alter biodiversity including local climatic effects, chemical, light and sound pollution and the introduction of non-native species. Thus, urbanisation impacts on moths need also to be considered in the context of the effects of climate, pollution and non-native species (see below).

Although reduced levels of moth abundance and diversity have long been associated with urbanisation (Taylor *et al*. 1978), there do not appear to have been any published studies of the specific impacts of urbanisation on the moth fauna of GB, nor of the relative value for moths of habitat fragments in urban surroundings compared with other degraded land-uses such as intensive agriculture. In California, Rickman & Connor (2003) found no consistent differences between leaf-mining moth communities of remnant habitats in urban versus agricultural settings.

Urban greenspace, including private gardens, supports diverse moth communities. As with agriculture, intensive management of gardens and parks (including pesticide use) is expected to reduce moth numbers, although quantitative studies are lacking. Recent trends for reduction in garden size, both in new-build developments and through in-fill (building new housing in existing gardens), and loss of vegetated area to hard surfaces (e.g. driveways, parking, patios, decking) and garden buildings (e.g. sheds, greenhouses) (Loram *et al*. 2008; Smith 2010; UK National Ecosystem Assessment 2011) will have reduced resources available to moths, but no population-level studies have been conducted.

In contrast, increased public awareness of biodiversity and interest in ‘wildlife gardening’ may have improved habitat quality in some gardens and parks, and the cultivation of non-native plants has provided opportunities for a few native and newly-colonising moth species (see section Non-native species).

*Habitat loss summary*. Direct evidence of the impact of historical habitat loss, decreasing quality or fragmentation on moth abundance or diversity is largely lacking. However, the weight of contemporary evidence suggests that reducing the intensity of agricultural management (including at field boundaries) and reinstating traditional management to recently neglected broadleaved woodlands increase moth abundance and diversity at the landscape scale. The implication is that the predominant trends in land-use management in 20th-century GB and concomitant loss of breeding habitat must have resulted in considerable declines for many moth species.

*Chemical pollution*

Eutrophication (increased soil and water fertility caused by unintended nutrient inputs from fossil fuel combustion and agriculture) is altering the plant composition and vegetation structure of many habitats, often in conjunction with other drivers such as management intensity and climate change (Bobbink *et al*. 1998; Van der Wal *et al*. 2003; Hartley & Mitchell 2005). Biodiversity of plant and insect populations (e.g. butterflies) correlates negatively with nitrogen input (Pollard *et al*. 1998; Stevens *et al*. 2004; Öckinger *et al*. 2006; WallisDeVries & van Swaay 2006), so there may be substantial, unquantified impacts on moth populations resulting from such chemical pollution.

Links between other forms of chemical pollution and moth populations appear completely unstudied in GB. It has been suggested that the population increases seen amongst moths that utilise lichens and algae as larval hostplants (e.g. the footman moths in sub-family Lithosiinae) might be linked to the recovery of some of these organisms following amelioration of sulphur dioxide pollution (Fox *et al*. 2006b). However, there is no direct evidence for such causality. Similarly, while there has been much research into the impacts of pollution by heavy metals and other chemicals on humans, other vertebrates and plants (e.g. Sharma & Agrawal 2005), there have been few studies involving moths. Negative fitness impacts of chemical pollution on moth larvae have been shown in Europe (Mitterböck & Fuhrer 1988; van Ooik *et al*. 2007; van Ooik & Rantala 2010), but population effects have not been established.

In summary, there is no evidence currently available to suggest that chemical pollution in its many, complex and interacting forms is a driver of change in moth populations in GB. However, as a key constituent of agricultural intensification and through negative effects on the insects themselves, larval hostplants and other essential resources, it is probable that chemical inputs in the form of herbicides, insecticides and fertilizers have contributed to the decline of GB moth populations.

### *Light pollution*

Many moth species are attracted to artificial light, although the mechanistic basis for this behaviour is not entirely clear (Young 1997). Artificial light elicits a wide range of responses in many animal and plant species, but there is insufficient knowledge about impacts in the wild, especially among invertebrates (Longcore & Rich 2004; Rich & Longcore 2006; Sutherland *et al*. 2006; Poot *et al*. 2008; Royal Commission on Environmental Pollution 2009; Stone *et al*. 2009; Bruce-White & Shardlow 2011).

Outdoor lighting can cause direct mortality, increase exposure to predators and have disruptive effects on various elements of moth behaviour and life cycles (Frank 2006; Bruce-White & Shardlow 2011). However, such effects vary between species, populations and even individuals, as well as with the spectral composition of the light sources. Furthermore, direct impacts of light pollution must be quantified separately from the other effects of urbanisation and habitat loss that usually accompany an increase in lighting levels.

Unfortunately, despite a massive increase in background light levels in GB and many other parts of the globe, there have been few studies on the impact of outdoor lighting on moths (e.g. Eisenbeis 2006; van Langevelde *et al*. 2011) and none that have assessed population-level or community-level effects.

Conrad *et al*. (2006) undertook a comparison of moth population trends from the RIS network using satellite data on the change in background illumination levels in GB. There was no significant difference between total moth abundance in areas exposed to increased background light levels and those unaffected. However, illumination data were available for only a short period (1992-2000), and therefore this finding does not preclude light pollution as a driver of long-term moth declines in GB.

In summary, although the attraction of moths to artificial light has been known for centuries and disruptive and fitness-reducing impacts of such attraction have been demonstrated, light pollution remains uninvestigated as a possible cause of population-level changes in moths.

*Climate change*

Climate change has already caused considerable modification of geographical range, abundance and phenology for many species globally (Parmesan & Yohe 2003; Gregory *et al*. 2009; Thackeray *et al*. 2010; Chen *et al*. 2011) and is perceived to be a major threat to biodiversity (Thomas *et al*. 2004a; Pounds *et al*. 2006; Thomas *et al*. 2006; Ohlemüller *et al*. 2008; Bálint *et al*. 2011; Maclean & Wilson 2011).

In GB (and elsewhere in north-west Europe), moderate levels of climate warming may bring opportunities for thermally-constrained species such as insects and there is strong evidence, for example, that some butterflies have already expanded their ranges and flight periods in response to climate change (Roy & Sparks 2000; Warren *et al*. 2001; Hill *et al*. 2002; Davies *et al*. 2006; Menéndez *et al*. 2007). At the same time, climate change may threaten other species through the loss of thermally suitable habitat space (Franco *et al*. 2006; Wilson *et al*. 2007; Maes *et al*. 2010), altered phenological synchrony with hostplants (Singer & Parmesan 2010) and even hybridization (Mallet *et al*. 2011).

Established links between climate change and the decline of moths in GB are limited at present. Population trends of a small group of northerly distributed species (i.e. those with a southern range margin within GB) decreased compared with southerly distributed moths (Conrad *et al*. 2004), and Morecroft *et al*. (2009) found significant decreasing population trends for moth species with more northerly European distributions at northern, upland sites in the UK Environmental Change Network.

In addition, several studies have found links between winter conditions and moth declines, indicative of climatic influence. Population levels of Garden Tiger correlate closely, and negatively, with winter precipitation and mean spring temperature, suggesting a link between climate change and the severe decline (89% decrease in population index,1968-2002) of this moth (Conrad *et al.* 2002). Furthermore, studies of moth declines in both GB and the Netherlands found significant relationships between overwintering life-cycle stage and species trend; moths that overwinter in the egg stage had declined (on average) more than others (Conrad *et al*. 2004; Groenendijk & Ellis 2011; and a similar result for butterflies in WallisDeVries & van Swaay 2006). Species overwintering as larvae or pupae had also decreased, while species that are adults during the winter had, on average, increased in both countries.

Another effect of winter and early spring climate has been observed on Winter Moth *Operophtera brumata* populations in the Netherlands. The synchrony of larval hatching date with the availability of its larval food resource (bud burst of *Quercus robur*) decreased over time, because of larvae hatching in advance of bud burst (Visser & Holleman 2001). The degree of synchrony was reduced by warmer spring temperatures combined with no change in the incidence of days with frost during the winter. Such asynchrony is predicted to cause a large increase in larval mortality, which is a major driver of population dynamics in this species. Thus, prolonged or high levels of asynchrony might cause population decreases in this moth species, although intense selection pressure to restore synchrony (or adaptive asynchrony) may rapidly redress this problem (van Asch *et al*. 2007; Both *et al*. 2009; Singer & Parmesan 2010).

In contrast, climate change is also expected to benefit elements of GB’s moth fauna. There is already some evidence for range expansion and increased abundance among southerly distributed moth species (i.e. those with a northern range margin in GB). Morecroft *et al*. (2009) found that species with the most southerly distributions at the European scale showed significant increases in population levels at 10 sites in the UK. The moth species with the greatest population increases in GB according to Conrad *et al*. (2006) also had increased distribution size, and the northern range margins of a sample of eight macro-moth species had shifted northwards considerably (mean 79.5km 10 year$^{-1}$ northward shift, 1982-2009), rivalling the largest equivalent results for butterflies and Odonata (Hill *et al*. 2002; Hickling *et al*. 2005; Fox *et al*. 2011b). This intimates that southern moths may conform to the general pattern of poleward range expansions recorded among other taxa in GB and globally (Hickling *et al*. 2006; Chen *et al*. 2011). The study by Salama *et al*. (2007) in central Scotland found that increasing moth diversity was positively correlated with mean annual temperature.

The absence of moth abundance decline in northern GB compared with significant decreases in southern GB appears to relate to a greater proportion of species with increasing population trends in the north (Conrad *et al*. 2006; Fox *et al*. 2006b; Scottish Government 2007). This pattern is consistent with poleward range expansion and increasing abundance of some moth species through northern GB in response to climate change. However, other factors, such as different patterns of land use and land-use change in northern GB, could equally be responsible.

Other generally positive climate change impacts on moths in GB include increased immigration (Sparks *et al*. 2005; Morecroft *et al*. 2009), colonisation (Parsons 2003, 2010) and phenological change. The latter includes many examples of advancement and increased duration of flight period and additional generations in apparent response to climate warming, both in GB and elsewhere in Europe (e.g. Fletcher 2006, 2009; Salama *et al*. 2007; Altermatt 2010; Pöyry *et al*. 2011).

In summary, although the evidence is limited at present, GB moths appear to be responding to climate change in qualitatively similar ways to butterflies. There are suggestions of climatic effects leading to the decline of some species, but also clear evidence of apparently positive impacts on species populations and distributions. Future climate change may, of course, alter this balance if new conditions are unsuitable for moth species in GB, plus the interaction between climate change and habitat loss, for example through sea-level rises, may damage specialist moth communities of coastal wetland habitats (e.g. Fisher's Estuarine Moth *Gortyna borelii*; Ringwood *et al*. 2004).

### *Non-native species*

Globally, non-native species are regarded as a principle driver of biodiversity decline and an ongoing threat to species and habitats (Mack *et al*. 2000; Manchester & Bullock 2000; Gurevitch & Padilla 2004; McGeoch *et al*. 2010). Many species of non-native plants, vertebrates and invertebrates are established in GB, and there are numerous negative impacts on native biodiversity (Brown *et al*. 2008; Lack 2010; Lever 2010; Holt *et al*. 2011).

There have been no quantitative assessments of the impact of non-native species on moth populations in GB. Nonetheless, negative effects might be expected via the influence of invasive plant species and introduced animals (e.g. deer) on habitat quality and larval hostplant resources. Examples of specific impacts include the invasion of semi-natural habitats of Slender Scotch Burnet *Zygaena loti*, Transparent Burnet *Z. purpuralis* and *Eudarcia richardsoni* by *Cotoneaster* spp. shrubs (M. Parsons & T. Prescott, pers. comm.). Experiments in the United States found that non-native woody plants supported significantly lower abundance and species richness of moth and butterfly larvae than native trees and shrubs, even if the alien plants were in the same genus as the native hostplants (Burghardt *et al*. 2010). The impact of new predators is even more poorly understood, with species such as Harlequin Ladybird *Harmonia axyridis* and the parasitic fly *Sturmia bella* spreading rapidly and having the potential to impact on moth populations as well as other insects (Brown *et al*. 2011; Gripenberg *et al*. 2011).

Set against these examples is the success of some colonising and rapidly increasing moths that utilise non-native plants as larval hosts (Parsons 2003, 2010; Conrad *et al*. 2006; Fox *et al*. 2011b). Blair's Shoulder-knot *Lithophane leautieri*, for example, utilises Cupressaceae trees and shrubs and, having become established on the south coast of GB in the mid-20th century, spread rapidly northwards (146km 10 year$^{-1}$, 1982-2009) and increased substantially in abundance (16.5% year$^{-1}$, 1968-2002). Other Cupressaceae-feeding moths show similar patterns, including recent colonists (e.g. Cypress Carpet *Thera cupressata* and Cypress Pug *Eupithecia phoeniceata*) and native species (e.g. Juniper Carpet *T. juniperata* and Juniper Pug *E. pusillata*). The latter moths were formerly restricted to semi-natural habitats where their only native larval hostplant Juniper *Juniperus communis* occurs but, in recent decades, both moths have colonised many gardens in which ornamental Cupressaceae species have been planted (Waring *et al*. 2009).

Non-native species have not been directly linked with moth declines or extinctions in GB as yet, though there is clear potential for negative impacts. On the contrary, non-native plants have enabled new moths to colonise GB and a few native species to extend their distributions.

*Exploitation of populations*

Collecting of wild specimens of macro-moths was once an integral part of the natural history study of this taxon in GB. In modern times, despite an increase in popular interest in macro-moths, collecting of specimens is less commonplace. Although over-collecting has often been postulated as a cause of decline or extinction for rare moths and butterflies in GB, there is little evidence to support the assertion (Young 1997; Asher *et al*. 2001), contrary to other taxa (Diamond *et al*. 1989; Roberts & Hawkins 1999; Jackson *et al*. 2001; Rosser & Mainka 2002; Dirzo & Raven 2003). Indeed, the large population sizes, phased emergence and short lifespan of many moth species also make it theoretically unlikely that anything but highly organised, exhaustive collecting could impact on any but the rarest localised species. Nevertheless, responsible collecting is strongly urged by relevant UK organisations, and there is a widely accepted code of conduct (Invertebrate Link 2002).

Young (1997) considered New Forest Burnet to be the only moth species for which there was credible evidence of extinction caused by collecting in GB. After discovery in 1869, nine sites were found in the New Forest in southern England, attracting large numbers of collectors, and the moth became extinct in 1927. The extinction proved short-lived, however, as another, isolated colony of the moth was later discovered in Scotland. The precise location of this remaining colony has not been publicised to reduce potential damage from collecting.

## Synthesis: why have GB moths declined?

Substantial decreases have occurred in overall abundance of macro-moths and the populations of many widespread species in GB and north-western Europe. In some cases, parallel reductions in distribution have been recorded (Conrad *et al*. 2002; Fox *et al*. 2011b). However, direct evidence to explain the trends is very limited. Correlative results and extrapolation from better-studied insect taxa (e.g. butterflies) provide the basis for our current understanding of the probable causes of moth declines and can be summarised as follows:

### *Multiple drivers of change*

This review indicates the influence of multiple drivers in the decline of GB moths. This is expected as it is improbable that each species in a diverse taxon would be affected by the same environmental and ecological factors. Various elements of habitat degradation, including habitat destruction, reduction in quality, loss of heterogeneity, and increased isolation, resulting from major land-use changes of the 20th century (agricultural intensification, changing woodland management, urbanisation) are very likely to have had an adverse impact on moths. For habitat specialist moths, this is a simple truth – the total area of semi-natural habitats such as unimproved calcareous grassland, heathland, fens and lowland raised bogs has decreased substantially. Generalist moths may also have been affected detrimentally by such losses but are, in addition, likely to have declined as changing land management (increased intensity in agricultural landscapes and a switch to high-forest silviculture) reduced available niches.

Research in agricultural and woodland settings show that moth abundance and species richness increase in response to techniques that reverse recent changes in management intensity. There is also correlative evidence that habitats subject to lower levels of management intensity change (e.g. organic farms) have higher abundance and species richness of moths.

Other drivers appear to be important too. There is strong evidence of both positive and negative climate change impacts. Currently, the impacts of chemical and light pollution and non-native species are insufficiently studied and understood to assess accurately. Thus far, most of the recognised impacts of non-native plants are positive, providing novel niches. Of the potential drivers of change considered in this paper, only direct exploitation of moth populations, in the form of collecting, is considered to be negligible in impact across the taxon.

### *Interactions and synergies*

Evidence from other taxa suggests that multiple drivers of population change are likely to interact, often in complex ways, and may produce synergies (Travis 2003; Brook *et al*. 2008). Thus, one driver, such as habitat loss, may act to reduce populations to levels where synergistic processes, both intrinsic (e.g. population dynamics, inbreeding depression) and external (e.g. other drivers such as climate change), and stochastic effects form amplifying feedback loops and drive species towards extinction. Such synergies have yet to be identified for moths in GB, but some have been elucidated for butterflies (e.g. interactions between habitat loss and the negative implications of isolation for populations, and between climate change and nitrogen pollution (WallisDeVries & van Swaay 2006; Bulman *et al*. 2007; Hanski & Pöyry 2007)).

The human activities that shape the environment tend to generate complex mixtures of change. For example, agricultural intensification causes habitat loss, but also changes spatiotemporal structure and heterogeneity, and chemical inputs alter botanical communities. Urbanisation also causes habitat loss, along with changes to the climatic environment, background lighting levels and chemical pollution. Isolating the relative contributions of these drivers to moth declines within the real world of human land use is an enormous challenge that has, as yet, received little attention.

## Future perspectives

Much moth research to date has focused on species that are economic pests on agricultural or forestry crops. The conservation biology of moths has been neglected as a research topic, particularly in comparison with butterflies and, as a result, although widespread declines of moth faunas have been identified recently from GB and other countries, knowledge of the underlying causes is scant. Fortunately, this has started to change. Ecologists are taking a greater interest in moths, spurred on by the pressing need to understand the causes and implications of biodiversity decline and the opportunities afforded by an ecologically-diverse and species-rich taxon supported by large surveillance and monitoring datasets. Sutherland *et al*. (2006) highlighted the need to understand the causes of moth declines as one of 100 ecological questions of high policy relevance, Butterfly Conservation continues to raise awareness of the declines among the public and policy makers alike (e.g. Fox *et al*. 2006b) and the UK Government added 71 species of widespread but rapidly-declining macro-moths to the UK Biodiversity Action Plan as Priority Species with the intention of stimulating research into causal factors and amelioration measures. The maintenance of recording and monitoring schemes gathering spatially extensive, long-term, time-series data on moths is vital to underpin future research and conservation.

It is hoped, therefore, that the next decade will see a continued surge in research interest leading to better comprehension of the changes taking place in GB's moth fauna. The following issues and questions are proposed to help understand and reverse the decline.

1. What is the complete picture of change for GB moths? Overall abundance has decreased, but the differing trends between northern and southern halves of GB provide a natural contrast that might shed light on the causes of change. Are the differences due to less-intensive land use and more extensive semi-natural habitats in northern GB or do they arise from climate change driving increases in range and abundance for southerly distributed moth species? Furthermore, population and/or distribution trends have been calculated for fewer than half of the c.900 macro-moth species and only a tiny proportion of the c.1600 micro-moths. Long-term distribution data have now been gathered by the National Moth Recording Scheme for all macro-moths in the UK (Fox *et al*. 2011b), and could be used to generate distribution trends and estimates of range margin shift. Revised national population trends from the ongoing RIS would also yield more up-to-date

information, and critical statistical analysis might yield further insight into the underlying causes. Trend analyses are currently impossible for all but a small minority of micro-moths (e.g. the Pyralidae), but greater co-ordination of micro-moth recording at the national level could generate suitable data in the medium term.

2. As agricultural intensification is considered to be a major driver of moth declines in GB, improved understanding of the impacts of different elements of agricultural management is required. Identification of the key factors that depress moth abundance and diversity would facilitate efforts to reverse the trends (e.g. through AES). For example, ‘What are the relative impacts of initial loss of habitat to cropped land versus the subsequent agricultural management?’, ‘How important is local habitat heterogeneity?’ and ‘What role do pesticides play in relation to other aspects of crop cultivation?’

3. More research is needed into land-management techniques that attempt to mitigate against biodiversity loss (Warren & Bourn 2011). If moth declines are to be reversed and wider biodiversity policy targets met, evidence-based AES prescriptions, woodland management practices and urban landscape designs are needed. Currently, there is little evidence that AES have benefited biodiversity, despite huge budget expenditure, at the national and European scale (Kleijn *et al*. 2011; but see Brereton *et al*. 2008). Crucially, the impact of such management techniques on populations is a vital but seldom addressed issue. Most studies, including those on moths, focus on recording changes in the abundance and species richness of adult animals in relation to management treatments and make no assessment of reproduction, immature stages or population dynamics (e.g. Feber *et al*. 1996; Pywell *et al*. 2004; Merckx *et al*. 2009b; Haaland *et al*. 2011). Management techniques may simply concentrate mobile adults within the landscape (e.g. at nectar resources) without contributing substantially to improved fitness or increased population levels. Worse still, interventions aimed at improving biodiversity might have a negative impact via source-sink effects (Severns 2011).

4. An equally critical question concerns the optimal targeting of AES for maximum benefit and cost-effectiveness. Theoretical and (limited) empirical evidence suggests benefits from clustering AES participation in the landscape (Merckx *et al*. 2009b; Gabriel *et al*. 2010), targeting extensively farmed land that retains relatively high levels of biodiversity (Kleijn *et al*. 2009) and, conversely, focusing on ‘simple’ landscapes where agriculture already dominates and semi-natural habitats are isolated (Tscharntke *et al*. 2005). An associated debate concerns the relative merits of setting land aside (or taking land out of cultivation) for biodiversity conservation (land sparing) versus reducing the intensity of agricultural management on farmland to benefit wildlife at the expense of production (land sharing) (Green *et al*. 2005; Hodgson *et al*. 2010). Apart from the recent work of Merckx *et al*. (2009b), there is no information on these contrasting strategies that relates directly to moths in GB.

5. The impact of outdoor, artificial lighting and background light pollution on moths and other nocturnal biodiversity is a topic requiring urgent ecological research (Sutherland *et al*. 2006). It is imperative that such studies aim to elucidate and quantify population-level effects and that research focuses on artificial lighting of types and intensities commonly experienced by wild moth populations. Does artificial light cause negative population-level effects in moth populations through increased mortality and disruption of life-cycles and behaviour? If so, what measures can be taken to reduce these impacts (e.g. through choice of lighting type, power, quantity and orientation, placement of lights and the periods that they are operated)?

6. Finally, although many impacts of climate change have been recorded for butterflies and other taxa in GB, little is known about the responses of moths to weather and climate (with the exception of Garden Tiger, Conrad *et al*. 2002). It would be insightful to assess the range margin shifts of all macro-moth species in GB, utilising the National Moth Recording Scheme database, and to attempt to relate shifts to climate change, habitat and larval hostplant distribution. In addition, the species richness of the macro-moth fauna in GB provides a good opportunity to detect poleward or uphill retreats

of high-altitude or northerly distributed species, which have proved rather elusive thus far.

The requirement for a research and conservation response elicited by the recently discovered widespread declines of moths in GB and beyond is substantial and challenging. These declines are one of the clearest signals yet of catastrophic biodiversity loss caused by anthropogenic environmental and land-use changes, which is of great conservation concern and threatens ecosystem services upon which the human race depends. Understanding and taking measures to reverse the declines of diverse insect faunas, such as GB macro-moths, are vital steps back from the brink.

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# Chapter 3: Long-term changes to the frequency of occurrence of British moths are consistent with opposing and synergistic effects of climate and land-use changes

Slightly modified from:
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## Abstract

Species' distributions are likely to be affected by a combination of environmental drivers. We used a dataset of 11 million species occurrence records over the period 1970-2010 to assess changes in the frequency of occurrence of 673 macro-moth species in Great Britain. Groups of species with different predicted sensitivities showed divergent trends, which we interpret in the context of land-use and climatic changes.

A diversity of responses was revealed: 260 moth species declined significantly whereas 160 increased significantly. Overall, frequencies of occurrence declined, mirroring trends in less species-rich, yet more intensively studied taxa. Geographically widespread species, which were predicted to be more sensitive to land-use than to climate change, declined significantly in southern Britain, where the cover of urban and arable land has increased. Moths associated with low nitrogen and open environments (based on their larval hostplant characteristics) declined most strongly, which is also consistent with a land-use change explanation. Some moths that reach their northern (leading edge) range limit in southern Britain increased, whereas species restricted to northern Britain (trailing edge) declined significantly, consistent with a climate change explanation. Not all species of a given type behaved similarly, suggesting that complex interactions between species' attributes and different combinations of environmental drivers determine frequency of occurrence changes.

Our findings are consistent with large-scale responses to climatic and land-use changes, with some species increasing and others decreasing. We suggest that

land-use change (e.g. habitat loss, nitrogen deposition) and climate change are both major drivers of moth biodiversity change, acting independently and in combination. Importantly, the diverse responses revealed in this species-rich taxon show that multifaceted conservation strategies are needed to minimise negative biodiversity impacts of multiple environmental changes. We suggest that habitat protection, management and ecological restoration can mitigate combined impacts of land-use change and climate change by providing environments that are suitable for existing populations and also enable species to shift their ranges.

## Introduction

The main drivers of global biodiversity change have been identified (Millennium Ecosystem Assessment 2005), but their impacts vary spatially, temporally and taxonomically. Drivers may also interact to produce synergistic or opposing effects (Travis 2003; Brook *et al*. 2008; Schweiger *et al*. 2010), but there are few empirical examples, particularly for insects, which comprise the majority of terrestrial biodiversity (Collen *et al*. 2012). Unquantified change and a resultant lack of evidence-based conservation, present pressing biological and strategic management challenges.

Here, we utilise a substantial dataset of species occurrence records to examine long-term changes of a species-rich insect taxon (Lepidoptera: macro-moths) in Great Britain (GB). Large-scale, comprehensive assessments of biodiversity changes in speciose insect taxa are rare (Thomas 2005; Mattila *et al*. 2008, 2009; Jeppsson *et al*. 2010). Moths constitute one of the largest groups of herbivorous insects, forming key links in food webs, inflicting damage (as well as pollination) on their plant hosts, and providing a major food source for insectivorous animals in many ecosystems (Strong *et al*. 1984).

We calculate long-term changes in frequency of occurrence of 673 lepidopteran species in GB and evaluate the trends in relation to species’ predicted sensitivities to recent climatic and habitat changes. Habitat modification, particularly agricultural intensification, is considered the pre-eminent cause of recent species declines in GB and other western European countries (Warren & Key 1991; Robinson & Sutherland 2002; Kleijn *et al*. 2009). In parallel, climate change is eliciting changes in the geographical range, abundance, phenology and biotic interactions of Lepidoptera species (Parmesan 2006). Climate change provides a shifting context for the impacts of habitat modification, either amplifying or ameliorating species' responses depending upon ecological traits and biogeographical situation.

Gradients of land use, climate and species' distributions combine conveniently to provide distinct (often opposite) predictions of changes to species' occurrence in GB. Northern GB retains a higher proportion of semi-natural habitats than southern GB, where levels of land conversion to intensive agriculture and urbanisation have been greater (Morton *et al*. 2011). Therefore, moth species that are not strongly constrained by climate and occur widely in GB, might be expected to decline in the south while remaining relatively stable in the north, in response to land-use changes. On the other hand, many insect species (including many macro-moths) reach the north-western climatic limit of their European range within southern GB. These species should benefit from climate change, leading to the opposite prediction – they should potentially increase as the climate has warmed (Hickling *et al*. 2006). In contrast, arctic-alpine species that are restricted to northern and montane areas in GB might be expected to decline in response to regional warming. By considering warm-adapted, cold-adapted and relatively climate-insensitive (within GB) species across a broad gradient of land-use intensity, we attempt to tease apart the effects of change in land use and climate on GB moths.

Land-use changes involve altered management (e.g. increased fertilizer input) as well as conversion from one land-use type to another. We considered these effects by analysing the occurrence changes of moths that are monophagous on larval hostplants that possess different environmental requirements. Trait-based analyses of plant trends have been linked to drivers of change (Carey *et al.* 2008), utilising Ellenberg indicator values to characterise the realized niches of plants along environmental gradients, such as those relating to soil chemistry and light availability (Ellenberg 1979). Thus, by considering the Ellenberg indicator values of moth larval hosts, we can examine links between drivers of botanical change and changes to the frequency of occurrence of moths.

Here, we test three hypotheses: (i) macro-moth species will show a wide diversity of changes as they respond to diverse drivers, but will have declined overall, mirroring wider biodiversity trends. (ii) The responses of species with different geographic distributions (southern, northern, widespread) are expected to differ because the effects of climate and land use may differ between these species categories. (iii) Moth occurrence trends will be associated with hostplant attributes (Ellenberg indicator values); specifically, moths that use types of plant that are in decline, such as those associated with low nitrogen soil conditions, will also be in decline.

We found support for each hypothesis, enabling us to assess long-term moth biodiversity change. These results will guide future research into drivers of biodiversity change and inform ecological management to buffer species from negative impacts.

## Materials and Methods

### *Data sources*

GB species occurrence records for macro-moths (here defined as Lepidoptera families: Hepialidae, Cossidae, Zygaenidae, Limacodidae, Sesiidae, Lasiocampidae, Saturniidae, Endromidae, Drepanidae, Geometridae, Sphingidae, Notodontidae, Erebidae, Nolidae and Noctuidae) for the period 1970-2010 were obtained from the National Moth Recording Scheme database: 11,074,870 records were extracted. These were collated from volunteer observers during recording for distribution atlases organised by the Biological Records Centre and Butterfly Conservation (Heath & Emmet 1983; Hill *et al*. 2010) (accessible via the National Biodiversity Network http://data.nbn.org.uk).

Interspecies detectability differences can be an issue with analysis of occurrence data (MacKenzie *et al*. 2006; Kéry, Gardner & Monnerat 2010), so we only considered within-species changes over time. New knowledge of species’ biology or novel collection methods may also alter detectability (Jeppsson *et al*. 2010). Thus, non-resident species and those subject to taxonomic revision since 1970 were excluded from the analysis. We also excluded species for which recording methodologies changed (e.g. most Sesiidae were excluded because the recent introduction of pheromone lures has greatly improved detection rates) and species that occurred in <10 grid squares in the 1970-1999 period, as no range margin could be determined for these species (see next section). This left 673 species (10,462,519 records in total) for our analysis.

Each species occurrence was attributed to a 10km x 10km grid square of the GB Ordnance Survey (OS) National Grid (hereafter ‘grid squares’) for analysis. The records cover 93% of GB grid squares.

### *Classification of southern, northern and widespread species*

Range margins were determined as the mean latitude of the 10 most northerly or southerly occupied grid squares in 1970-1999 (Hickling *et al*. 2006), the baseline period for our analysis. Species were then classified into three groups, based on the 488km North gridline (OS National Grid). ‘Southern species’ had a northern (cold) range margin that occurred in the southern half of GB (i.e. south of 488km North OS). ‘Northern species’ had a southern (warm) range margin north of 488km North. ‘Widespread species’ did not meet either criteria, occurring in both northern and southern GB (Fig. 1). There was little evidence of taxonomic bias between these groups (Fig. A3.1).

### *Analysis of changes in frequency of occurrence*

Temporal and spatial variation in recording intensity (Boakes *et al*. 2010) must be accounted for in analyses of species occurrence data (Ponder *et al*. 2001; Hedenäs *et al*. 2002; Telfer *et al*. 2002; Hassall & Thompson 2010; Pardo *et al*. 2013). We interpreted moth occurrence data using the program Frescalo to determine temporal trends for each species (Hill 2012). This method utilises the presence or absence of ‘benchmark’ species to assess recording intensity at a given location. A local set of benchmark species was defined for each (focal) grid square, based on species occurrence data in surrounding ‘neighbourhoods’. The fraction of benchmark species observed in a focal square enables recording effort to be estimated, which can then be used to adjust the observed frequencies of species occurrence. The adjusted frequencies are then used to assess trends over time (see Hill 2012 and Appendix 3 for detailed explanation).

Frescalo was applied to the total moth dataset (673 species), split into two time periods of roughly equal numbers of records, 1970-1999 versus 2000-2010. For each time period, a grid square was categorised as having species detected (1) or not-detected (0) (giving a sample of 720,969 data points). Neighbourhoods were defined based on spatial proximity and floristic similarity using 1970 onwards vascular plant data from Preston *et al*. (2002). For each location in our analysis, the corresponding neighbourhood was defined as the 50 most floristically similar (using a spatial smoothing kernel) grid squares selected from the 100 geographically closest squares to each location (Appendix 3).

Change in moth species' frequency of occurrence was estimated by considering the relative reporting rate (RRR; Appendix 3) of each species in each time period (1970-1999 and 2000-2010) (Hill 2012). Temporal trends for each species were expressed as the yearly change in RRR, calculated as the overall change between the mid-points of the two time periods (i.e.1984 and 2005 respectively) divided by the number of intervening years. The significance of these trends was determined using a z-test by:

$$z = \frac{t_2 - t_1}{\sqrt{\sigma_1{}^2 + \sigma_2{}^2}}$$

where $t_1$ and $t_2$ are the relative reporting rates of a given species from the first and second time periods and $\sigma_1{}^2$ and $\sigma_2{}^2$ are the variances associated with the RRR for periods $t_1$ and $t_2$ respectively. Trends in RRR were determined to be significant (at the 95% confidence level) if $|z| > 1.96$. The analyses of Frescalo trends were carried out in R v2.9.2 (R Development Core Team 2009).

Finally, for widespread species, RRR trends were recalculated separately for the northern and southern halves of GB, dividing the data along the 488km North gridline.

### *Correlation with host plant and environmental variables*

We tested host plant effects for the subset of 56 GB macro-moths that are monophagous (Skinner 2009; Waring *et al*. 2009) on vascular plant species for which distribution and trait (Ellenberg indicator values) data were available.

Long-term GB distribution changes of the plants (1930-1960 versus 1987-1999) and Ellenberg values were derived from PLANTATT (Hill *et al*. 2004). We used all Ellenberg values in PLANTATT (soil nitrogen, soil pH, soil moisture and shade tolerance) excluding salt tolerance, for which there was insufficient variation for the plants in our analysis.

We tested whether changes in frequency of occurrence ($\Delta$RRR year$^{-1}$) of the 56 moth species were correlated with distribution change of their hostplants. We fitted a multiple regression of moth changes against their host's Ellenberg values for light, moisture, reaction (pH) and nitrogen. In all these statistical models, we included species distribution grouping ('southern' or 'widespread' species; no northern species were part of the monophagous group) as a control variable. Regressions were fitted in R with moth $\Delta$RRR year$^{-1}$ as a response variable and either plant distribution change or Ellenberg traits as explanatory variables. Initially, model residuals did not conform to normality, so three outlying data points were removed to rectify this (Shapiro test for normality of residuals: W = 0.9776, $p = 0.42$, n = 53), although results were qualitatively similar when including these data. We considered the phylogenetic non-independence of species by fitting a mixed effects model with genus and family as random effects. Higher-level phylogenetic relationships are not well resolved in Lepidoptera so a full comparative analysis using a phylogeny was not possible (Mutanen *et al*. 2010). We used the *lme4* and *lmerTest* packages (Bates *et al*. 2008; with significance of variables assessed using Satterthwaite's approximation for degrees of freedom, Kuznetsova *et al*. 2013).

## Results

British macro-moth species decreased significantly in frequency of occurrence between the periods 1970-1999 and 2000-2010 (Wilcoxon signed-rank test on $\Delta$RRR year$^{-1}$ using all species: V = 87558, n = 673, $p < 0.001$): 260 of the 673 species exhibited significant declines ($p < 0.05$), with a further 157 species showing a tendency to decline. In contrast, 160 species increased significantly ($p < 0.05$) in frequency of occurrence, with 96 others showing a tendency to increase. Thus, 420 (62%) of the species have undertaken significant changes in frequency, with 1.6 times as many decreasing as increasing (Table A3.1). The magnitude of these changes was relatively similar between groups (median

ΔRRR $year^{-1}$ for significantly increasing species = 0.006 [range 0.002-0.033]; significantly declining species: median = -0.006 [range = -0.024 - -0.002]; Table A3.1). The results reveal a wide diversity of occurrence changes among moths.

Geographically limited species showed contrasting trends (Fig. 1). Species restricted to northern GB (trailing edges of distributions) declined significantly in frequency of occurrence (with 94% of species declining; V = 10, n = 17, $p$ = 0.002). In contrast, species confined to southern GB did not show a significant change overall (V = 8575, n = 186, $p$ = 0.87): 24% of species declined significantly while 27% increased significantly.

On average, geographically widespread species decreased in frequency of occurrence (V= 39066, n = 470, $p$ < 0.001; Fig. 3.1): 45% of individual species in this group declined significantly. When trends for widespread species were recalculated separately for southern and northern GB, we found disproportionately larger declines in the south (Fig. 3.2). There was no significant change in frequency of occurrence of widespread species in northern GB (V = 53569, n = 470, $p$ = 0.55), but a significant decline in the south (V = 37017, n = 470, $p$ = < 0.001).

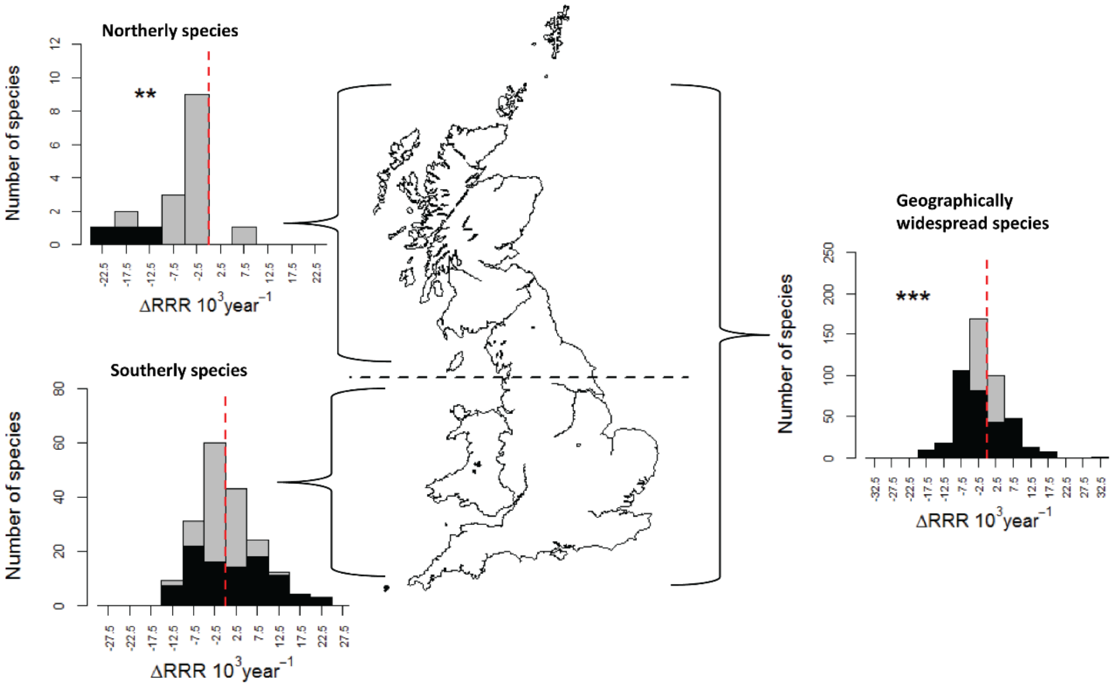

**Fig. 3.1** Change in frequency of occurrence (per year change in relative reporting rate, RRR) 1970-1999 versus 2000-2010 for southerly distributed, northerly distributed and geographically widespread moths. Significant results shown as ** $p < 0.01$ and *** $p < 0.001$. Species with individually significant changes ($p < 0.05$) are shown in black. Change values are multiplied by $10^3$ to improve axis legibility.

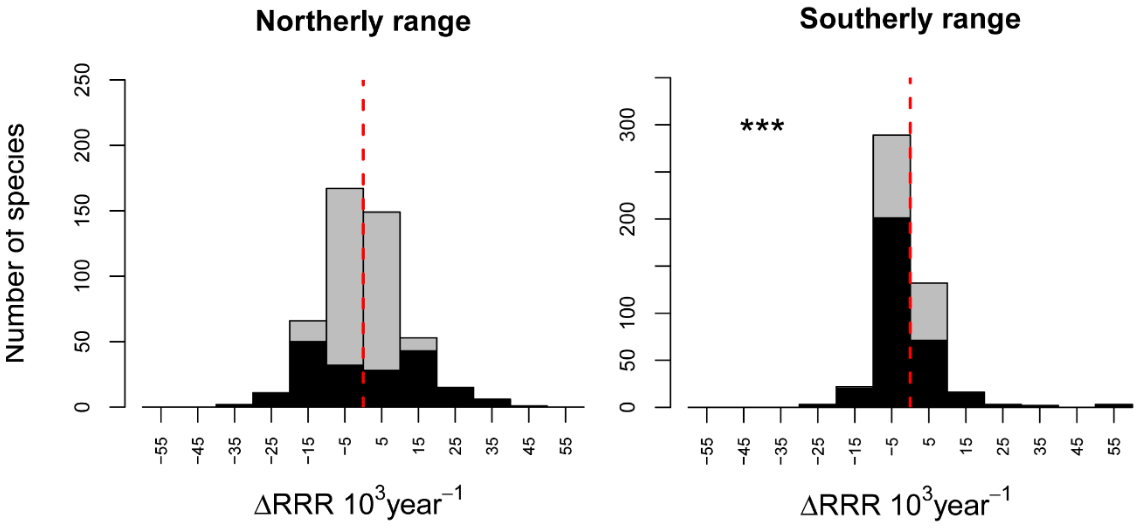

**Fig. 3.2** Change in the frequency of occurrence (per year change in relative reporting rate, RRR) 1970-1999 versus 2000-2010 of geographically widespread moth species in the northern and southern halves of GB (divided by

488km North OS gridline, see Fig. 3.1). Species with individually significant changes ($p < 0.05$) are shown in black. Change values are multiplied by $10^3$ to improve axis legibility.

Changes in frequency of occurrence of monophagous macro-moths and distribution changes of their larval hostplants were not significantly linked (linear regression: slope = 0.002, t = 1.33, $p = 0.19$, $R^2 = 0.03$; mixed model: slope = 0.002, t = 1.99, $p = 0.057$; n = 53 species for both; Fig. A3.2). However, there was a negative relationship between moth species' trends and their hostplant Ellenberg light values and a positive correlation between moth trends and host Ellenberg nitrogen values (Table 3.1; Fig. 3.3). Moths utilising larval hostplants growing in open, low-fertility conditions declined over time compared to species using plants in more shaded, nitrogen-rich environments. There were no relationships between moth trends and Ellenberg values for moisture or reaction.

**Table 3.1** Relationships from a multiple regression and linear mixed model of host plant Ellenberg indicator values on change in frequency of occurrence of monophagous moth species (n = 53 for both). Significant results ($p < 0.05$) shown in bold text. Species distribution grouping (Distribution) ('southern' or 'ubiquitous' species; no northern species were part of the 53 species) was included as a covariate, with the intercept representing southern species.

| | Model 1 multiple regression | | | | Model 2 mixed effects (phylogenetic control) | | | |
|---|---|---|---|---|---|---|---|---|
| **Coefficient** | **Coefficent** | **SE** | **t** | ***p*** | **Coefficent** | **SE** | **t** | ***p*** |
| Intercept | 0.0057 | 0.0050 | 1.14 | 0.261 | 0.0042 | 0.0049 | 0.849 | 0.401 |
| Light | -0.0014 | 0.0005 | -2.64 | **0.011** | -0.0011 | 0.0005 | -2.179 | **0.035** |
| Moisture | -0.0007 | 0.0006 | -1.20 | 0.236 | -0.0006 | 0.0005 | -1.139 | 0.261 |
| Reaction | -0.0004 | 0.0005 | -0.89 | 0.378 | -0.0007 | 0.0005 | -1.432 | 0.160 |
| Nitrogen | 0.0013 | 0.0006 | 2.32 | **0.025** | 0.0015 | 0.0005 | 2.772 | **0.008** |
| Distribution | 0.0006 | 0.0012 | 0.52 | 0.607 | 0.0006 | 0.0012 | 0.477 | 0.636 |

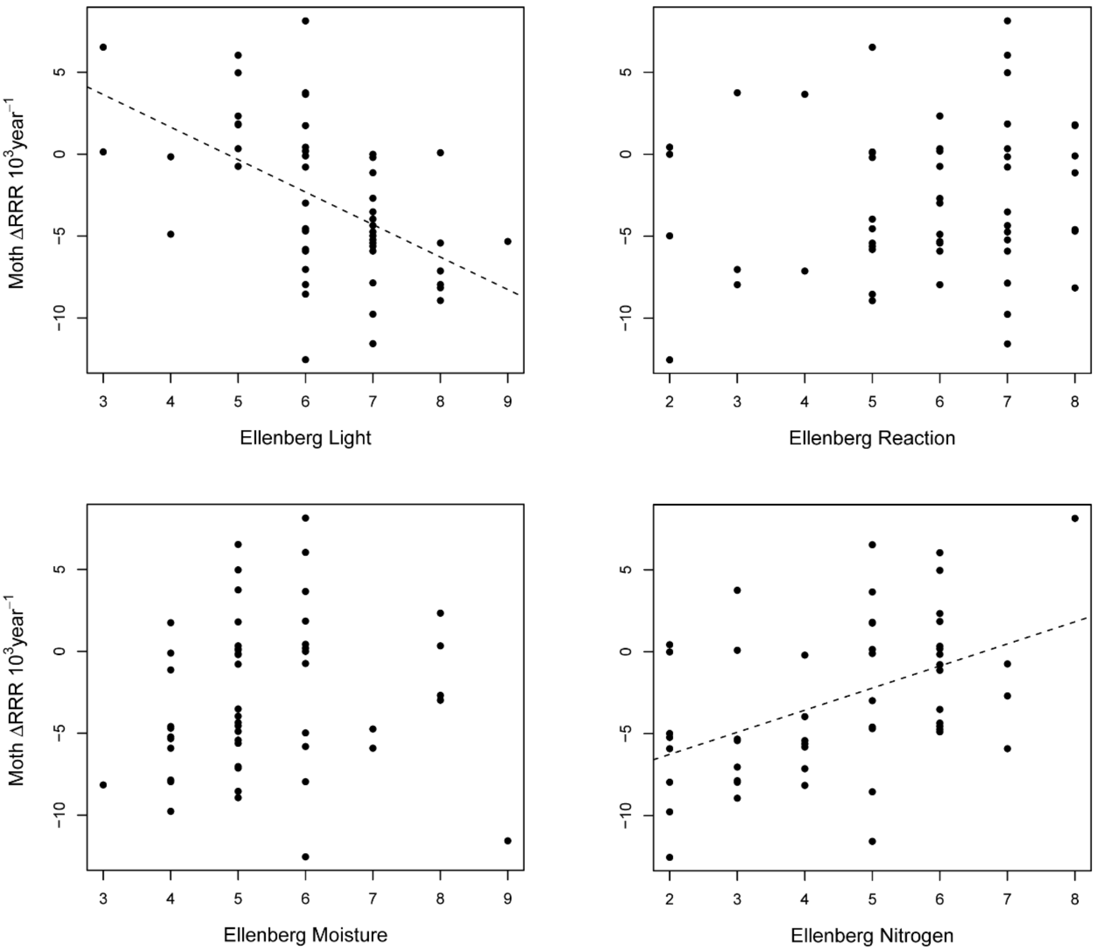

**Fig. 3.3** Change in the frequency of occurrence (per year change in relative reporting rate, RRR) 1970-1999 versus 2000-2010 of monophagous moth species in relation to host plant Ellenberg indicator values. Change values are multiplied by $10^3$ to improve axis legibility. Dashed lines are from univariate regressions.

## Discussion

Macro-moth species in GB decreased overall in frequency of occurrence between 1970-1999 and 2000-2010, in keeping with a significant decrease in GB macro-moth abundance over a similar period (Conrad *et al.* 2006), moth distribution trends in other countries (Mattila *et al.* 2008; Groenendijk & Ellis 2011) and declines in other insect taxa (Warren *et al*. 2001; Cameron *et al*. 2011). It provides further evidence that invertebrates are as negatively impacted by environmental change as vertebrates (Thomas *et al*. 2004; Collen *et al*. 2012). The diversity of trends suggests that combinations of different drivers are resulting in a mixture of responses.

The occurrence trends were calculated using the Frescalo method to control for spatiotemporal variation in recorder effort (Hill 2012). Without controlling for this bias, variation in the intensity of recording can confound assessments of species occurrence over time. The method estimated frequency of occurrence, which is a function of both local abundance and distribution extent (Appendix 3, Fig. A3.3, A3.4).

The Frescalo method makes a number of assumptions. One is that the probability of finding a species in a locality can be estimated by its frequency in the neighbourhood (floristically similar grid squares in close spatial proximity). We believe this is reasonable because moth species tend to be associated with specific ecotypes and plant communities and because plant communities are generally good indicators of a range of local environmental conditions (e.g. soil structure, pH, moisture levels and microclimate; Ellenberg 1979). A second potential consideration of the Frescalo method is that poorly recorded neighbourhoods cannot provide information about local species frequency. This was not an issue in the current analysis of moth data at 10km resolution with neighbourhoods of 50 grid squares, but it could be if analyses were conducted at finer spatiotemporal scales. Finally, the Frescalo method may have limited applicability for less speciose taxonomic groups that have few potential benchmark species.

Our results demonstrate different patterns of change in the frequency of occurrence among macro-moths with different geographical distributions and hostplant traits, providing full or partial support for each of our hypotheses. Moths as a whole decreased in frequency of occurrence, as did northern and geographically widespread species, while southerly distributed species showed no overall trend. Additional analyses showed that geographically widespread species only decreased in the southern half of GB and showed no overall trend in the north. Correlations between trends of monophagous moths and Ellenberg indicator values of their hostplants revealed mixed findings.

The development of an understanding of the drivers of moth biodiversity change in GB is a vital step for conservation biologists and practitioners. We propose an interpretation of our findings based on two major drivers of change for GB biodiversity: habitat modification and climate change. There is growing indirect evidence of the impacts of these drivers on GB moths (Merckx *et al.* 2012; Fox 2013), but we acknowledge that other factors may be involved and drive changes in the occurrence of individual species.

The overall decrease in moth frequencies, and that of the subset of geographically widespread species, is consistent with a response to high levels of habitat modification, as for butterflies (Warren *et al*. 2001), although it does not exclude other explanations.

Our second set of hypotheses related to the performances of three geographically defined groups of moths. Southerly distributed (warmth associated) species were predicted to increase in response to regional climate warming (Fig. A3.5), but they also inhabit the parts of GB with the highest levels of land-use change. Some of these species increased and others decreased (resulting in no overall significant trend in this group, Fig. 3.1). This might reflect a diversity of habitat and climatic sensitivities, although such results could also be due to the species being insensitive to recent changes in climate and land-use.

In northern GB, cold-adapted species have declined; a response consistent with synergistic negative effects of climate change and habitat modification (as found for four northern GB butterfly species, Franco *et al*. 2006). This is in keeping with other studies implicating climate change in the retraction of warm range margins of cold-adapted Lepidoptera (Thomas *et al*. 2006; Chen *et al*. 2011; Dieker *et al*. 2011). Specific conservation measures may be required for these trailing edge populations (Hampe & Petit 2005), including steps to minimise negative land-use impacts and the protection of climatic refugia.

Geographically widespread species only decreased, on average, in southern GB; population monitoring has yielded similar findings (Conrad *et al*. 2006; Fox *et al*. 2011). Almost all of the widespread species also occur in warmer parts of Europe, and are unlikely, therefore, to have experienced a climatic deterioration of conditions in southern GB, although there may be exceptions (e.g. Garden Tiger *Arctia caja* Conrad, Woiwod & Perry 2002) due, for example, to local climatic adaptation. A greater proportion of widespread species is increasing in northern GB (Fig. 3.2) perhaps reflecting the positive impacts of climate change for some species.

Southern GB has undergone greater loss of semi-natural habitats since the early 20th century than the north. Comparison of 10km grid square resolution land cover data for 1931-1941 with 2000 data suggests an increase in arable and urban land of 20% and 6%, respectively, in southern GB, and a 4% decrease of arable and 1% increase in urban land in the north (T. Jucker pers. comm.; Jucker 2010). Although these habitat conversion trends have slowed recently, the overall pattern of greater habitat modification in the south has been retained and ongoing degradation in habitat quality (e.g. loss of botanical species richness in linear features) has been recorded (Haines-Young *et al*. 2003; Carey *et al.* 2008). We suggest that the decline of widespread moth species in southern GB is predominantly linked to habitat modification. Further research is needed to assess whether these rates of decline will cause regional extinctions, and to identify effective conservation strategies in the wider countryside (Kleijn *et al.* 2011).

The variation among species is as revealing as the overall trends (Table A3.1). Sixteen of the 17 northern species showed a declining trend, suggesting relatively consistent responses to drivers of change. In contrast, many southern species increased significantly while others decreased significantly; a pattern also seen among widespread species. Given that species vary in their habitat associations and likely responsiveness to different elements of climate, it is not surprising that simultaneous habitat and climatic changes generate increases in frequency in some species and declines in others (Menéndez *et al*. 2007).

Much recent research has focussed on species' traits as predictors of biodiversity decline (Mattila *et al*. 2008; Öckinger *et al*. 2010), but success in explaining climate change responses has been limited (Angert *et al*. 2011). We examined traits of the plant hosts of moths, which are expected to reflect sensitivity to land-use changes more than the climate (Firbank *et al*. 2008; Kleijn *et al*. 2009).

Surprisingly, we found no significant relationship between changes in hostplant distributions and frequency of occurrence of dependent moths (Fig. A3.2). However, specialist moths rarely occupy the entire range of their larval hosts (Quinn *et al*. 1997), and change in hostplant distribution might occur in parts of the range unoccupied by the associated moth. In addition, thresholds of hostplant abundance, quality and local distribution may determine moth persistence (Menéndez & Thomas 2000), but these are not accounted for in assessments of distribution change. Finally, the lack of association may stem from the inherent differences in the measures being compared (frequency of occurrence change for moths versus distribution change for plants).

We did find significant correlations between changes in the frequency of occurrence of moth species and Ellenberg values of hostplants for two predictors, showing that monophagous moths that utilise plant species associated with high light intensity and low-fertility soils tended to decrease most strongly (as have plants with these traits, Carey *et al.* 2008). Decreases among plants and their specialist herbivores associated with open, nutrient-poor conditions can be attributed to habitat modification directly, through changing agricultural and woodland management, and also indirectly, for example due to eutrophication of the environment (Warren & Key 1991; Firbank *et al.* 2008; Kleijn *et al*. 2009; Payne *et al.* 2013). Such impacts, mediated through botanical communities (Payne *et al*. 2013), have rarely been recorded among herbivores (Hendriks *et al*. 2013). Although enrichment may be reversible on individual sites, new approaches to the management of nutrients in the wider countryside will be required to address declines of species restricted to low nutrient environments (Robertson & Vitousek 2009).

Synergistic climate change interactions, both negative and positive, may also occur. Warmer conditions extend the growing season (Menzel & Fabrian 1999) leading to increased plant growth, particularly if coupled with rising soil fertility. Thus, climate change could favour shade-tolerant species and could, perversely, reduce warm microclimatic niches required by invertebrates (WallisDeVries & van Swaay 2006; Oliver *et al.* 2012). On the other hand, for moth species that utilise plants favoured in high-nitrogen environments, eutrophication may facilitate climate-driven range expansion (Betzholtz *et al.* 2012).

Understanding species' responses to the drivers of biodiversity change is vital to develop adaptive conservation strategies (Mawdsley *et al*. 2009). The diverse patterns of change revealed by our study suggest that drivers of trends are likely to differ between species, necessitating multifaceted approaches to conservation. Nevertheless, a generic solution is to maintain existing high-quality habitats and create new areas (Lawton *et al*. 2010). This will minimise declines (e.g. of widespread species in the south) and maximise increases (e.g. of southern species), regardless of whether species are responding most strongly, or in combination, to land-use or climatic changes. Hence, conservation strategies should aim to retain sufficient quantity and quality of habitat to minimise negative synergistic effects (Oliver *et al*. 2010; Araújo *et al*. 2011), while facilitating the exploitation of opportunities created by climate warming (Hodgson *et al*. 2011; Thomas *et al*. 2012). This requires the protection of remaining habitats from deleterious impacts, but also sufficient knowledge of land management techniques to maximise habitat quality. Such knowledge is limited for moths but can start by identifying landscape elements and management practices associated with enhanced species richness and abundance (Fuentes-Montemayor *et al*. 2011; Merckx *et al*. 2012).

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## Chapter 4: Using citizen science butterfly counts to predict species population trends

Slightly modified from:
Dennis EB, Morgan BJT, Brereton TM, Roy DB & Fox R (2017) Using citizen science butterfly counts to predict species population trends. *Conservation Biology* **31**, 1350–1361.

### Abstract

Citizen scientists are increasingly engaged in gathering biodiversity information, but trade-offs are often required between public engagement goals and reliable data collection. We compared population estimates for 18 widespread butterfly species derived from the first four years (2011-2014) of a short-duration citizen science project (Big Butterfly Count, BBC) with those from long-running, standardized monitoring data collected by experienced observers (UK Butterfly Monitoring Scheme, UKBMS). BBC data are gathered during an annual three-week period, whereas UKBMS sampling takes place over six months each year.

An initial comparison with UKBMS data restricted to the three-week BBC period revealed that species population changes were significantly correlated between the two sources. The short-duration sampling season rendered BBC counts susceptible to bias caused by inter-annual phenological variation in the timing of species' flight periods. The BBC counts were positively related to butterfly phenology and sampling effort. Annual estimates of species abundance and population trends predicted from models including BBC data and weather covariates as a proxy for phenology correlated significantly with those derived from UKBMS data.

Overall, citizen science data obtained using a simple sampling protocol produced comparable estimates of butterfly species abundance to data collected through standardized monitoring methods. Although caution is urged in extrapolating from this UK study of a small number of common, conspicuous insects, we found that mass-participation citizen science can simultaneously contribute to public engagement and biodiversity monitoring. Mass-participation citizen science is not an adequate replacement for standardized biodiversity

monitoring but may extend and complement it (e.g., through sampling different land-use types), as well as serving to reconnect an increasingly urban human population with nature.

## Introduction

Citizen science, the participation of members of the public in gathering research and monitoring data, is increasing rapidly across many scientific disciplines, including biodiversity conservation (Dickinson *et al*. 2012; Follett & Strezov 2015). Public involvement in biodiversity recording and monitoring has a long history in some countries (Miller-Rushing *et al*. 2012; Pocock *et al*. 2015). Distinction can be made, however, between citizen science projects in which standardized protocols are used to conduct systematic, repeatable sampling in long-term studies (e.g. the Breeding Bird Survey; Gregory & Baillie 1998) or for hypothesis-driven enquiry (e.g. Conker Tree Science; Pocock & Evans 2014) and schemes reliant on opportunistic sampling undertaken with relatively unstructured protocols (e.g. eBird; Sullivan *et al*. 2009). Opportunistic schemes with simple sampling protocols reduce barriers to participation (e.g. time commitment, prior knowledge) and may thus engage large numbers of new, inexperienced citizen scientists. Although these increase sample size and public outreach, the data gathered may lack credibility (Riesch & Potter 2014; Lewandowski & Specht 2015). Standardized schemes may have much greater barriers to participation and therefore rely on fewer dedicated, skilled volunteers. However, the abilities of these participants to undertake biodiversity monitoring may be comparable with those of professional scientists (Chase & Levine 2016). Biodiversity citizen science projects often involve trade-offs between the goals of public engagement and education (counteracting the extinction of experience; Soga & Gaston 2016) and the collection of reliable data for research (Chase & Levine 2016; Lakeman-Fraser *et al*. 2016).

Many aspects of citizen science biodiversity research have been examined, including the quality of observations (Lewandowski & Specht 2015), participants' motivations (Hobbs & White 2012), and the development of new data-analysis techniques (Bird *et al*. 2014). However, few studies have compared population trends based on relatively unstructured sampling undertaken by mass-participation citizen science with those derived from long-term systematic monitoring and none, to our knowledge, involving terrestrial invertebrates. We derived and compared species population trends from two contrasting citizen science projects in the United Kingdom (UK) - the Big Butterfly Count (BBC) and UK Butterfly Monitoring Scheme (UKBMS).

The BBC is an annual survey of widespread butterfly species launched in 2010 that encourages participation by members of the general public (www.bigbutterflycount.org). It seeks to engage people with little or no experience with biodiversity monitoring and aims to enhance public awareness and interaction with nature and to gather species-abundance data. To minimize barriers to participation, the sampling protocol is simple: 15-minute counts of 18 butterfly species and two diurnal moths over three weeks in the summer. Consequently, and thanks to a high media profile, BBC has met its aims of mass-participation (mean = 47,636 people involved per year 2013-2015) and raising awareness but, given the target audience, likelihood of identification mistakes, and simple method, counts may not provide a meaningful indication of butterfly population change.

The UKBMS, initiated in 1976, has a robust, standardized recording protocol in which weekly fixed-route counts are conducted over six months each year at >1,000 sites. High levels of commitment and identification skills are required so participants tend to be experienced amateur butterfly observers or professional conservationists, and the high-quality data generated are used to produce population trend estimates for 56 of 59 regularly breeding UK butterfly species, as biodiversity indicators by government (Brereton *et al*. 2011a; Eaton *et al*. 2015), and in scientific research (e.g. Dennis *et al*. 2013; Oliver *et al*. 2015b; Thackeray *et al*. 2016). We tested the validity of BBC data for estimating species trends by determining whether population changes derived from BBC data were comparable with those from UKBMS.

Butterfly abundance differs throughout the year as one or more broods emerge. These phenological patterns vary year to year in response to the weather (Sparks & Yates 1997) and show long-term trends due to climate change (Roy & Sparks 2000). Because the BBC runs for just three weeks each summer, inter-annual variation in counts for each species may result from differing phenology rather than real population changes. We assessed temporal variation in phenology with respect to the BBC survey period to determine its influence on estimates of annual change. Furthermore, we investigated whether population-change estimates from the BBC, in conjunction with weather covariates, can provide an accurate indicator of how populations are faring. In the rapidly expanding field of citizen science, we sought to provide a rare test of the validity of a mass-participation approach to biodiversity monitoring.

## Methods

### *Big Butterfly Count*

The BBC runs annually in late July and early August during the peak in overall abundance of butterflies. In 2010, the scheme ran for nine days. Since 2011, the BBC occurs over a period of up to 24 days each year (Table A4.1), although participants can additionally submit counts taken throughout July and August. Due to this difference, we excluded 2010 data from analyses and used BBC data from 2011 to 2014. Participants count 18 widespread butterflies (Table A4.2) and two day-flying moths for 15 minutes during bright weather. No training is provided, sightings are submitted online, and minimal verification of sightings is undertaken. Counts can be undertaken anywhere in the UK. If counting from a fixed position, the maximum number of each species seen at any time is recorded rather than an additive total so as to reduce double counting. BBC data are summarized in Table A4.3 and show the scheme's rapid growth. Sightings are spatially referenced and land-use type is recorded by the participant. The majority of counts are taken in gardens (65% on average, Table A4.4). An average of 12%, 11%, and 4% are taken in fields, other rural, and woodland sites, respectively, and a small number are taken in other land-use types.

### *UK Butterfly Monitoring Scheme*

The UKBMS counts are undertaken along line transects, typically 2-4km, with systematic, standardized methods (Pollard & Yates 1993). In 2014, 1,223 UKBMS transects were monitored (Brereton *et al*. 2015). Counts can be made throughout the main season for UK butterfly activity; the core period is April-September. A 5m wide fixed transect route is walked weekly at specified times of the day and weather conditions, and all butterflies seen are identified and counted. In practice approximately 30% of core-season weekly counts are missed (Dennis *et al*. 2013). Transect counts are used to generate annual indices of relative abundance from which population trends can be calculated.

*Comparisons of BBC and UKBMS data*

We compared species abundance estimates from the two schemes in three ways. First, we examined agreement through direct comparison of annual growth rates. Second, we investigated the effects of sampling effort and phenology. Finally, we tested whether UKBMS trends may be predicted over 36 years (1980-2015) and 10 years (2006-2015) based on BBC data and an appropriate weather variable acting as a proxy for butterfly phenology.

The BBC and UKBMS are inherently different, independent datasets, and although sample locations are self-selected by participants in both schemes, the representation of habitats may differ. Overall UK coverage of each scheme is shown in Fig. A4.1. Most BBC counts are undertaken in gardens, whereas UKBMS locations are biased toward semi-natural habitats that are often managed to benefit biodiversity (Brereton *et al*. 2011b). We compared the habitats covered by the schemes by summarizing land-cover data from 2007 (Morton *et al*. 2014) in the 1km squares sampled in each scheme. For each UKBMS transect, the central 1km x 1km grid square was used to characterize the habitat.

*Comparison of annual growth rates*

To make an initial direct comparison between the two schemes, we limited the UKBMS data to counts made within the BBC survey period each year and restricted the analysis to the 18 butterfly species counted by the BBC (Table A4.2). Because BBC data are available for only a three-week period, by initially restricting the UKBMS data to the same period we could directly compare the two schemes in the absence of seasonal differences, for example due to multiple broods (which are sampled by the UKBMS).

Following Roy *et al*. (2015), we determined annual population growth rates for each species from the two datasets. In brief, we defined $\mu_{i,t}$ as the expected total count of a species at site *i* in year *t* across *vi,t* visits, and regarded this as the realisation of a Poisson random variable. Annual proportional changes in abundance were assumed to be the same across sites, such that we estimated annual growth rate (*Rt*) as

$$R_t = \log\left(\frac{\mu_{i,t+1}/v_{i,t+1}}{\mu_{i,t}/v_{i,t}}\right) \qquad (1)$$

which leads to

$$\log(\mu_{i,t}) = \sum_{j=1}^{t-1} R_j + \log(\mu'_{i,1}) + \log(v_{i,t}) \qquad (2)$$

where $\mu'_{i,t} = \mu_{i,t}/v_{i,t}$. Standard generalized linear model (GLM) software, for example in R (R Core Team 2016), may be used to fit this model. However, the many sites represented in the BBC data each require the estimation of a site parameter each year; hence, the model described is computationally challenging to fit to BBC data with standard GLM software because of the amount of computer memory required. Therefore, we adopted a concentrated (or profile) likelihood approach (Morgan 2008; Pawitan 2013) that reduces the number of parameters to estimate and results in efficient model fitting (Dennis *et al*. 2016).

With the notation $S_i = \log(\mu'_{i,1})$, apart from an additive constant, the log-likelihood may be written as

$$l = \mathrm{Log}(L) = \sum_{i=1}^{S} \sum_{t=1}^{T} \left[-\exp\left\{\sum_{j=1}^{t-1} R_j + S_i + \log(v_{i,t})\right\} + y_{i,t}\left\{\sum_{j=1}^{t-1} R_j + S_i + \log(v_{i,t})\right\}\right]. \qquad (3)$$

Then for site *i* we obtain

$$\frac{\partial l}{\partial S_i} = \sum_{t=1}^{T} \left[-\exp\left\{\sum_{j=1}^{t-1} R_j + S_i + \log(v_{i,t})\right\} + y_{i,t}\right], \qquad (4)$$

and equating to zero gives

$$S_i = \log\left\{\frac{\sum_{t=1}^{T} y_{i,t}}{\sum_{t=1}^{T} v_{i,t} \exp\left(\sum_{j=1}^{t-1} R_j\right)}\right\}. \qquad (5)$$

Substituting Eq. (5) into Eq. (3) results in a concentrated likelihood that can be maximized simply with respect to {*R j* }. We maximized the likelihood with the *optim* function in R and the BFGS algorithm (Nocedal & Wright 1999).

We estimated the net change, *N*, over *T* years for each survey with

$$\widehat{N} = \sum_{t=1}^{T} \widehat{R}_t, \qquad (6)$$

where the variance of $\widehat{N}$ is the sum of all the entries of the covariance matrix for the growth rates. We adjusted for overdispersion by scaling standard errors with the square root of the ratio of the Pearson chi-square statistic to its degrees of freedom.

### *Effects of phenology and effort*

Seasonality of life-cycle phenology results in differences in counts of adult butterflies throughout the year and complicates the analysis of population data (Rothery & Roy 2001; Dennis *et al*. 2013, 2016). We used UKBMS data to establish how the BBC data were influenced by changes in flight-period phenology. Seasonal abundance patterns for each species in each year were estimated by fitting an appropriate generalized abundance index model (GAI) (Dennis *et al*. 2016) to the UKBMS data (without date restriction, in contrast to the comparison of annual population growth rates). For univoltine and bivoltine species, a phenomenological GAI is based on the assumption that the flight period of each brood follows a normal distribution ($\mu$, mean flight date; $\sigma$, standard deviation). For species with complex seasonal flight patterns, which are difficult to model parametrically, a GAI was fitted using a spline to describe the seasonal variation. The approach used for each species is in Table A4.2.

For each univoltine and bivoltine species, we plotted the total BBC count per day and the estimated annual seasonal pattern from the UKBMS GAI. The BBC counts from all dates were used, rather than only the official three-week sampling period. We explored the relationship between BBC data and sampling effort and phenology. For each species, a negative-binomial model with log link was fitted using the glm.nb function from the MASS package (Venables & Ripley 2002) in R. The response was the total BBC count per day, and measures of effort (log counts per day) and phenology based on the estimated seasonal pattern from the UKBMS were covariates. We also modelled the number of counts per day (rather than the total BBC count); however, this measure was right skewed and therefore less satisfactory. The estimated seasonal pattern from the GAI (which sums to unity across the season) formed the measure of phenology for a given day and year. This is in anticipation of positive associations between BBC count and both sampling effort and the timing of sampling coinciding with the peak in species' seasonal patterns.

### *Predicting UKBMS species trends from BBC data*

We assessed whether UKBMS species' population trends were described by the BBC data with weather covariates as a proxy for phenology. We used a simple linear model to regress UKBMS abundance indices for 2011-2014 on BBC data and weather covariates and the index for the previous year (autoregression) to account for potential density dependence.

We used a GAI to estimate UKBMS indices. In a given year, the GAI produces a relative abundance, *Ni*, for each site *i* (Dennis *et al*. 2016). Given the variation in UKBMS sites between years, we fitted a Poisson GLM with year and site factors and used scaled predicted year effects as indices of abundance (Dennis *et al*. 2013).

We used BBC data from the official three weeks of sampling as a covariate in the linear model; the sum of the total counts per day was scaled by daily effort (defined as the log of the number of counts for all species for that day). However, scaling by the numbers of counts produced similar results.

Average monthly mean temperatures (Parker *et al*. 1992) and total rainfall (Alexander & Jones 2000) for central England for spring (March-May) and summer (June-August) were used as weather covariates. All weather covariates were standardized to have zero mean and unit variance. The maximum correlation between weather covariates was 0.67.

Potential longer-term (rather than for 2011-2014 only) effects of weather and density dependence were accounted for by fitting a linear model to the GAI index values for 1980-2014; the index values in the previous year and the four weather covariates were explanatory variables. The products of the slope coefficients and covariates from each model were included as optional offsets in the linear models to allow for potential longer-term effects than those for 2011-2014 only.

We used the dredge function in the MuMIn package (Barton 2016) in R to select models based on the Akaike information criterion (AIC). Given the few years for which BBC data were available, we allowed up to two variables only and only one weather covariate (either as a covariate for 2011-2014 or as an offset for weather from 1980 to 2014). The relative importance of the BBC and weather covariates was assessed using the relaimpo package (Grömping 2006) in R.

Each year UKBMS data are collated (from online and hard copy sources) and verified. Unverified UKBMS data were available for 2015 online; hence, a GAI was fitted to incorporate these data and estimate an index of abundance for 2015. We compared this 2015 index, estimated from observed UKBMS data, with the abundance index predicted from the BBC linear model with the lowest AIC. An abundance index for 2015 was also predicted for each of the candidate models, and we assessed the model with the prediction closest to the index from the observed UKBMS data.

Population trends were compared by fitting linear models to the index of abundance, where the index for 2015 had either been estimated from UKBMS data or predicted from the best linear model. We estimated percent change over two periods (long-term for 1980-2015 and short term for 2006-2015) and calculated percent change with respect to the previous year. In doing so we assessed whether predicting the 2015 index from the BBC affected the overall UKBMS trend estimates.

## Results

### *Comparison of BBC and UKBMS data*

A greater proportion of 1km squares sampled in the BBC were classified as urban than were transects in the UKBMS (Table A4.5). This was expected given that most BBC counts were undertaken in gardens. The UKBMS squares contained a greater proportion of broadleaf woodland than the BBC, but the two schemes showed similar coverage of arable farmland and improved grassland.

*Comparison of annual growth rates*

There was a significant correlation between net species population changes from the two schemes for 2011-2014 ($\rho = 0.84$, $p < 0.001$) (Fig. 4.1). There was also a significant correlation ($p < 0.01$) between each of the year-to-year changes (Fig. A4.2). From 2011 to 2014, 11 of the 18 species had significantly positive and three had significantly negative change in abundance in the BBC, whereas 11 species had significantly positive and six had significantly negative change in the UKBMS. The remainder showed nonsignificant trends (Table A4.6). Population changes estimated from the two schemes were similar, although the BBC growth rates were less precise and tended to underestimate UKBMS growth rates. Changes were generally of a similar magnitude and were always of the same sign, with the exception of Comma *Polygonia c-album* and Small White *Pieris rapae*, and in no cases were the changes significantly different from zero and in opposite directions (Table A4.6). Nevertheless, there were significant differences in net change 2011-2014 between the two schemes for 11 species, and confidence intervals for BBC results were on average twice the width of the UKBMS results (0.38 and 0.19 respectively). Estimates of overdispersion were greater than unity for both schemes (Table A4.7). The BBC confidence intervals narrowed in 2013-2014 (average width 0.18) relative to 2012-2013 (0.38) because of the increasing number of counts (Table A4.3).

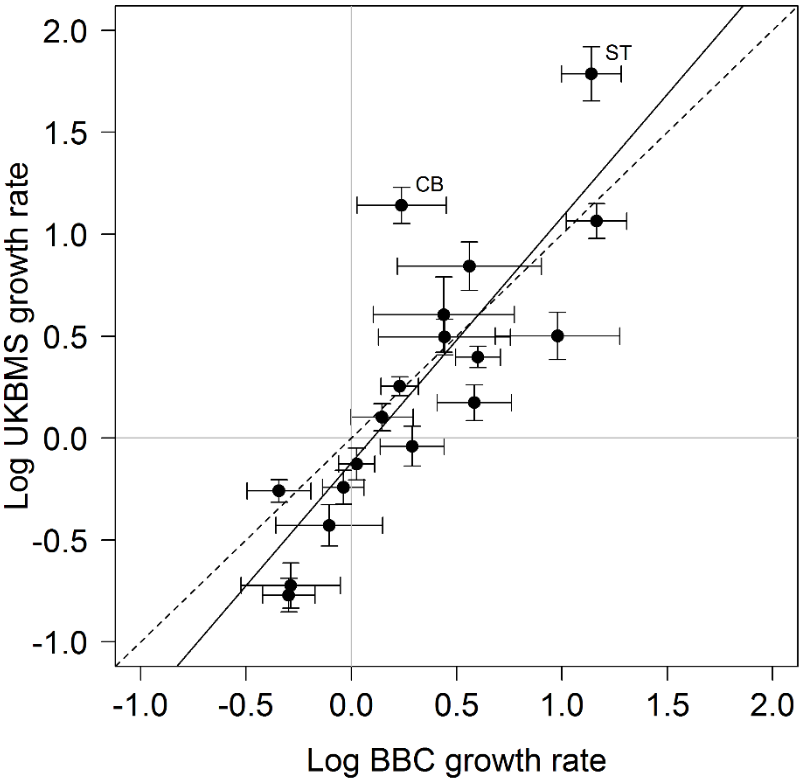

**Fig. 4.1** Comparison of estimated log growth rates of populations of 18 butterfly species from Big Butterfly Count (BBC) and UK Butterfly Monitoring Scheme (UKBMS) counts for 2011-2014 (error bars, 95% confidence intervals; solid grey lines, zero growth; dashed line, equal growth rates between the datasets; solid black line, fitted linear regression between the growth rates based on BBC and UKBMS data). The Small Tortoiseshell *Aglais urticae* (ST) and Common Blue *Polyommatus icarus* (CB) have the greatest differences.

*Effects of phenology and effort*

Overlaying total daily abundance of each species from BBC counts with phenology information from the UKBMS, revealed how BBC population estimates may be influenced by inter-annual variation in the timing of species' flight periods (examples in Fig. 4.2 & Fig. A4.3). For Gatekeeper *Pyronia tithonus* the peak flight period was fairly central in the BBC recording period in 2011 and 2013 but fell at the end of period in 2012 and near the beginning in 2014. For Large White *Pieris brassicae* timing of the second brood varied; in 2012 in particular, the peak fell outside the BBC period.

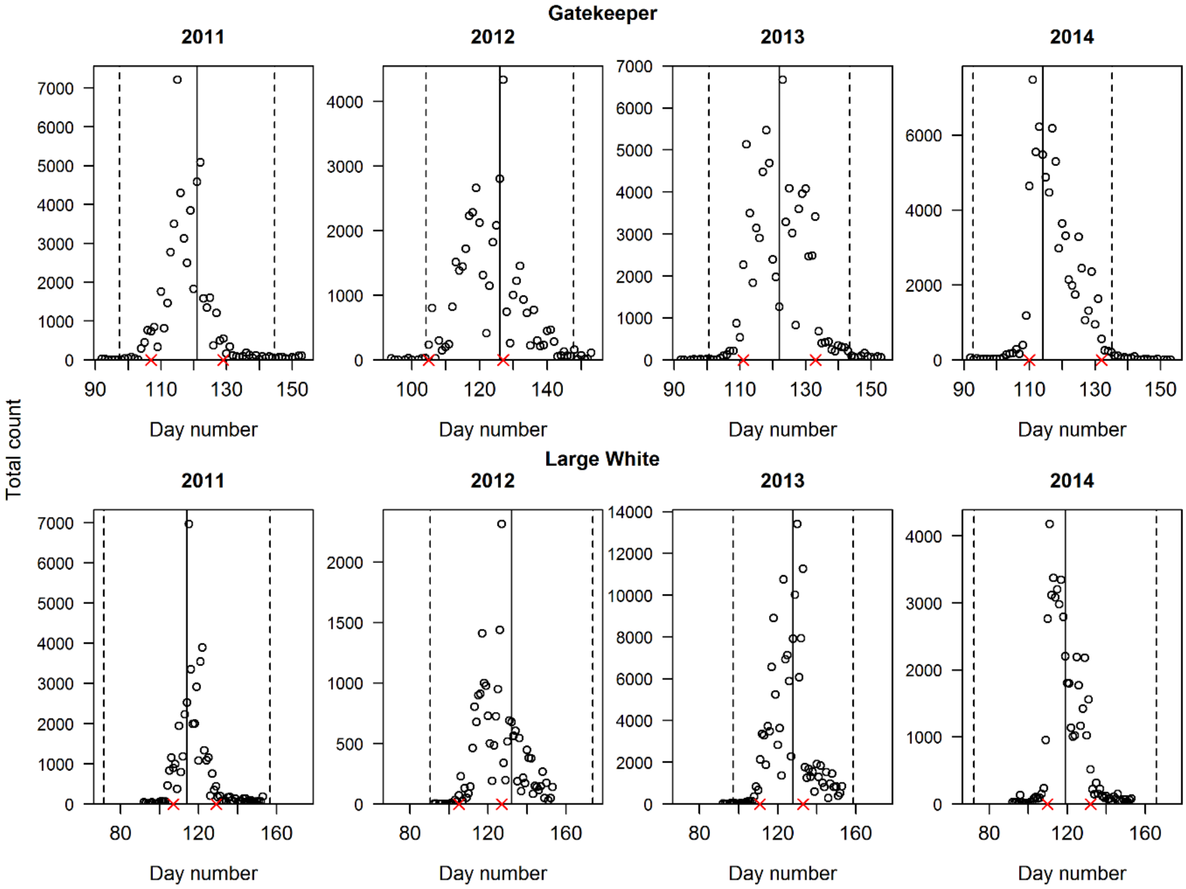

**Fig. 4.2** Total counts of Gatekeeper and Large White from Big Butterfly Count (BBC) data per day in each year, where day 1 is 1 April (vertical lines, mean flight dates estimated from a generalized abundance index model; dashed lines, twice the SD; top, 1st brood; bottom, 2nd brood; red crosses, official BBC survey period for each year).

Regressing the BBC counts on measures for effort and phenology showed good agreement between the counts and expected values, given the simplicity of the model used (Fig. 4.3 & Fig. A4.4). Residual deviance values suggested a good fit for the negative binomial model compared with the Poisson model (Table A4.8).

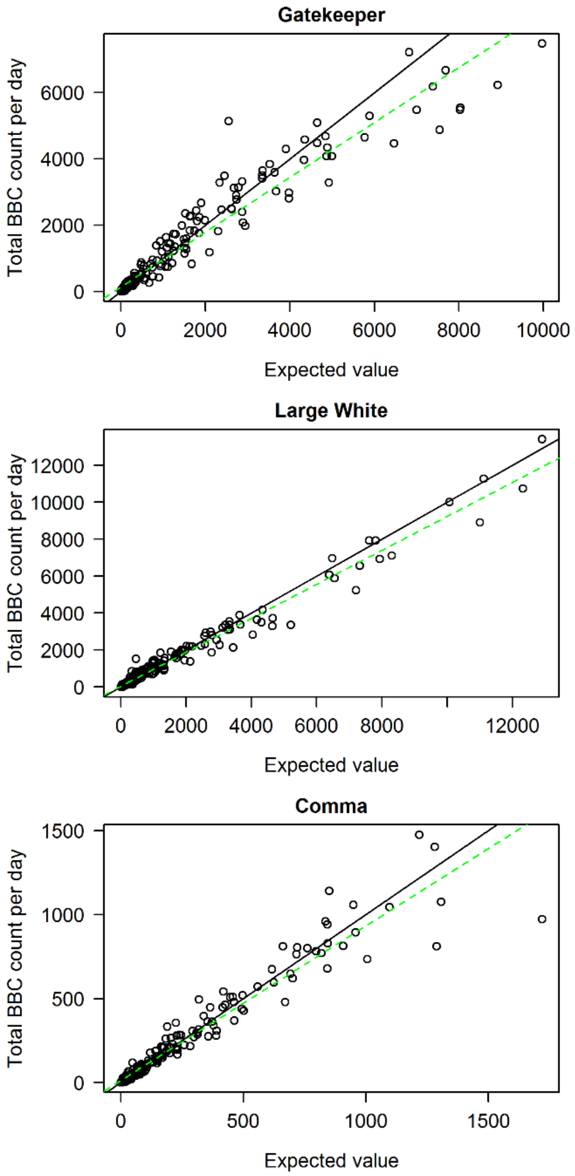

**Fig. 4.3** Total counts of three butterfly species (Gatekeeper, univoltine; Large White, bivoltine; Comma, multivoltine) from Big Butterfly Count (BBC) data per day versus the expected value from a negative-binomial model with log-link in which the response variable is the total count per day and measures of effort (log number of counts made) and phenology (from the corresponding generalized abundance index model curve) are covariates (black line, equal expected values and total counts; green dashed line, fitted linear regression through the points).

*Predicting UKBMS species trends from BBC data*

The BBC was a covariate in the best model (in terms of AIC) for 13 of 18 species (Table 4.1), in conjunction with summer rainfall, spring temperature, and spring rainfall each for three species; summer temperature for two; and offset long-term spring rainfall and autoregression for one species each. Of the 11 species where BBC and a weather covariate were in the best model, the relative importance of BBC exceeded the weather covariate for eight species (Table A4.9). For five species, BBC was not included in the best model, but autoregression was important. The observed 2015 index of abundance was within the 95% confidence interval of the best model for 10 out of 18 species, and only four species showed major discrepancies (Fig. 4.4).

**Table 4.1** Estimated population trends (percent changes) in relative abundance for 18 UK butterfly species for the best models and selected covariates in terms of Akaike information criterion (AIC) or predicted index closest to the observed 2015 UKBMS index, relative to observed UKBMS populations trends estimated from the generalized abundance index model. SPRt = spring temperature, SPRr = spring rainfall, SUMt = summer temperature, SUMr = summer rainfall, auto = auto-regression, of = offset variable. * = significant at $p < 0.01$.

| | Best fit | | 1980-2015 | | 2006-2015 | |
|---|---|---|---|---|---|---|
| Species | AIC | Prediction | Observed | Best AIC | Observed | Best AIC |
| Brimstone | bbc+SPRt | SUMr+of(auto) | 35.9 | 35.1 | -0.5 | -2.2 |
| Comma | bbc+SUMr | bbc+SUMr | 10.9 * | 10.9 * | -5.4 | -5.5 |
| Common Blue | bbc+SUMr | bbc+SUMr | -9.0 | -8.9 | 5.2 | 5.5 |
| Gatekeeper | bbc+SPRt | bbc+SUMr | -12.5 * | -12.4 * | -1.7 | -1.3 |
| Green-veined White | auto+SUMr | bbc+SPRt | -4.1 | -2.8 | 6.6 | 11.2 |
| Holly Blue | bbc+of(auto) | bbc+of(SUMr) | 4.1 | 1.4 | -6.1 | -14.1 |
| Large Skipper | auto+SPRt | auto+SPRt | -12.9 * | -13.5 * | -13.5 | -15.6 |
| Large White | bbc+SPRr | SPRt | -7.5 | -7.0 | -3.9 | -2.3 |
| Marbled White | auto+SUMr | auto+SUMt | -0.7 | 1.3 | 10.2 | 17.3 |
| Meadow Brown | bbc+SPRt | SUMt+of(auto) | -4.7 | -4.9 * | 2.4 | 1.7 |
| Painted Lady | bbc+SUMr | bbc+SUMt | -0.3 | -4.3 | -36.9 | -46.0 |
| Peacock | auto+SPRt | auto+of(SPRt) | -1.0 | 3.0 | 6.5 | 20.3 |
| Red Admiral | auto+SPRt | auto+bbc | 13.9 | 14.9 * | -11.9 | -9.3 |
| Ringlet | bbc+SUMt | of(SPRr) | 12.3 * | 11.3 * | 7.9 | 5.2 |
| Small Copper | bbc+of(SPRr) | SUMt+of(auto) | -14.2 * | -12.6 * | -11.8 | -6.1 |
| Small Tortoiseshell | bbc+SPRr | of(SPRr) | -27.0 * | -27.4 * | 30.2 * | 27.7 |
| Small White | bbc+SPRr | bbc+of(auto) | -7.2 | -6.2 | 0.1 | 3.3 |
| Speckled Wood | bbc+SUMt | bbc+of(SPRr) | 8.2 * | 8.0 * | 0.6 | 0 |

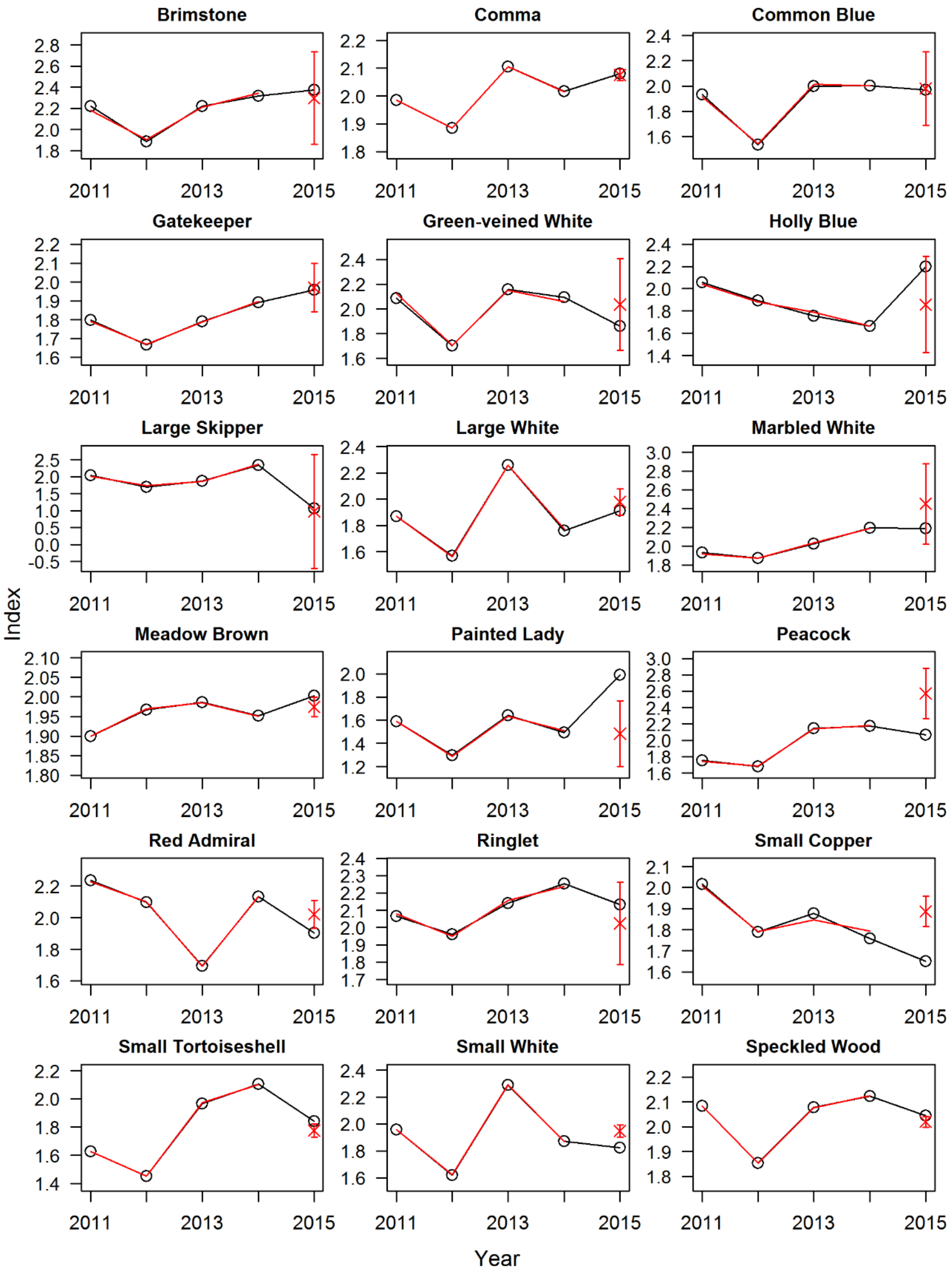

**Fig. 4.4** Comparison of the generalized abundance index from UK Butterfly Monitoring Scheme (UKBMS) data (black) and predicted butterfly abundance indices from the best model in terms of Akaike information criterion (red) (vertical line, 95% confidence intervals for the 2015 prediction).

There were significant correlations between estimated population trends (Fig. 4.5), where the values for 2015 were from the observed data or predicted from the best model: $\rho = 0.99$ for 1980-2015, $\rho = 0.95$ for 2006-2015, $\rho = 0.75$ for 2014-2015, where all $p < 0.001$. For 1980-2015, the difference between the two trends was < 5% for all species. For 2006-2015 and 2014-2015, the difference was < 5% for 13 and 10 species, respectively, out of 18. Significant trends were correctly identified for the seven species with significant UKBMS trends for 1980-2015, although two further species were predicted to have significant trends. There was greater correlation between the trends when the model with the best 2015 prediction was used (Fig. A4.5, Fig. A4.6).


1980-2015

2006-2015

**Fig. 4.5** Comparison of linear trends in relative butterfly abundance from the generalized abundance index model. The indices for 2015 are from observed data or predicted from the best model in terms of Akaike information criterion (solid grey lines, 0% change in relative abundance; dashed line, equal population trends). Abbreviations are for species common names (Table A4.2).

## Discussion

Citizen science appears to offer opportunities for largescale, cost-effective biodiversity monitoring. However, the reliability of species trends may be compromised in citizen science projects that prioritize public outreach goals because there is often a trade-off between mass participation and scientific rigor.

This reliability has rarely been tested empirically by comparing opportunistic citizen science data with standardized sampling data. Munson *et al*. (2010) found that eBird transect checklists predict bird species occurrence almost as accurately as highly standardized North American Breeding Bird Survey data. In contrast, Snäll *et al*. (2011) reported only weak overall correlation between opportunistic bird reports in Sweden and annual count data from a standardized transect-style survey. In the only terrestrial invertebrate examples we are aware of, Warren *et al*. (2001) and Oliver *et al*. (2015a) found correlations between UK butterfly species' occurrence trends assessed with opportunistic recording-scheme data and UKBMS population trends.

Population change estimates from the BBC and UKBMS using only counts from the official three-week BBC period were significantly correlated ($\rho$ = 0.84). This compares favourably with the value of 0.75 obtained by Roy *et al*. (2015) when they compared population trends from the UKBMS with the Wider Countryside Butterfly Survey, in which a reduced-effort UKBMS sampling protocol is used in randomly selected locations (Brereton *et al*. 2011b).

The temporal distribution of BBC counts showed a potential mismatch with annual phenological variation, and the BBC data were well described by measures of recording effort and phenology. Simple annual proportional changes in abundance calculated from the BBC could result from varying phenology and effort rather than true population changes and may mask or falsely predict declines and increases. This demonstrates that the results of snap-shot citizen science biodiversity projects, which often take place at fixed points during the year, are vulnerable to bias from temporal factors that are not normally measured in such projects, as well as from variation in participation.

Despite the limited number of years and lack of standardization or verification, linear models based on BBC data and simple weather covariates were surprisingly successful at predicting the UKBMS abundance index for 2015 and consequently correcting for the effects of changing phenology. The BBC was an important variable for 13 out of 18 butterfly species, and the difference between the two trends was < 5% for all species in 1980-2015. Predictions of population trends were good even for species that are not straightforward to identify for inexperienced participants (e.g. three *Pieris* species: Large, Small and Green-veined White). The significant correlation and similar estimates of population trends between the two schemes validates the use of BBC data in assessing abundance change for these UK butterfly species. We used only four years of BBC data; over time one would expect even better predictions from BBC.

Species with the poorest model predictions of the 2015 abundance index, and consequently greatest differences in trend estimates relative to the UKBMS, tended to be those recorded in fewer locations by the BBC. Wider confidence intervals for the prediction of the 2015 index were also associated with species recorded in fewer BBC locations. Species may be less well recorded by the BBC due to reduced population densities in locations such as gardens, where most counts are undertaken. This may be addressed by encouraging BBC observers to sample other land-use types. Population trends for some species may also be better described by alternative climatic covariates. For example, trends for migratory Painted Lady *Vanessa cardui* and Red Admiral *V. atalanta* may be better explained by weather from parts of their ranges outside the UK.

This study concerns only 18 widespread butterfly species in the UK; therefore, caution should be applied in extrapolating our conclusions to other taxa and areas. Relative to many invertebrate taxa, butterflies are conspicuous and popular, and, in the context of butterfly monitoring, the UK benefits from low species richness, high human population density, and a tradition of amateur natural history recording.

From a biodiversity conservation perspective, the limitations of BBC relative to the UKBMS are clear. The UKBMS provides population trends for all but one of the threatened butterfly species on the British Red List (18 of 19 species), whereas BBC primarily counts just 18 common butterfly species (all also monitored by the UKBMS). Even in the UK, mass-participation citizen science is unlikely to provide reliable data on the large number of threatened, habitat-specialist invertebrates.

Nevertheless, the BBC data, as validated by our results, provide the potential for additional or improved assessments of biodiversity change. For example, there is increasing interest in the biodiversity of urban areas, both as potential refuges for species whose habitats have been degraded in intensively farmed countryside and for the opportunities it affords for human-wildlife interactions and associated human well-being (Goddard *et al*. 2010; Shanahan *et al*. 2015). Sampling protocols developed for use in semi-natural habitat or open countryside may not be easily implemented in built areas and private gardens. The BBC samples more urban habitat than the UKBMS, and the majority of counts are undertaken in private gardens; hence, the BBC could provide a new biodiversity indicator for the performance of butterfly populations in gardens and parks, providing a valuable tool to engage the public and managers of urban greenspace.

The sampling of private gardens and urban areas as part of BBC also provides potentially useful population data for common butterfly species to complement UKBMS sampling of semi-natural habitat and the farmed landscape. While not of highest conservation priority, trends of common species are, nevertheless, of considerable interest due to the significance of such species to ecosystem function (Gaston & Fuller 2008). In the UK, the overall abundance of widespread butterflies decreased by 25% over 40 years (Fox *et al*. 2015), and many widespread species have significant negative population trends in the UK and the Netherlands (Van Dyck *et al*. 2009). Currently, the drivers of these declines are poorly understood. The BBC and UKBMS data could be combined in an integrated analysis (Pagel *et al*. 2014) representative of a wider range of land-use types, although variation in the scale and accuracy of the two surveys would need to be addressed, for example, by weighting different likelihood components (Francis 2011).

In practice, the financial costs of mass-participation citizen science versus standardized monitoring are an important factor, particularly where a new scheme is to be implemented. Both schemes incur considerable annual expenditure due to the essential involvement of professional staff, but the cost of running BBC is about a quarter that of the UKBMS. Aside from minor coordination, the primary cost of BBC arises from the need for media promotion to engage the public. Despite a larger overall cost due to greater coordination needs, it could be argued that the UKBMS is more cost-effective because data are collected for many more species, including those that are the main focus of conservation. Both schemes also require an online data system, however, as the primary monitoring method for UK butterflies, the UKBMS incurs additional costs associated with data validation, which is not undertaken in the BBC.

The UKBMS operating costs are contingent on the assumption that an adequate network of skilled, trained volunteers already exists or can be mobilized quickly. Without this, the start-up costs and lead-in time for a monitoring scheme would be substantially greater than for mass-participation citizen science, for example, if paid professionals were required (Carvell *et al*. 2016). As we have shown with the BBC, mass-participation citizen science may, in some instances and with suitable adjustments (e.g., for effort and phenology), provide meaningful estimates of population trends for common, easily identifiable species. Even if this is not the case (or cannot be tested), by raising awareness and providing informal education, citizen science projects may provide a means to develop the necessary pool of skilled, engaged volunteers to enable the establishment of standardized biodiversity monitoring of additional areas and of taxa that are not currently well-monitored.

Despite relatively simplistic modelling and only a few years of available data, and contrary to the scepticism with which mass-participation citizen science is sometimes viewed, we found that BBC can produce population change estimates for common butterflies comparable to standardized monitoring data collected by skilled recorders. These results establish BBC as an example of a citizen science win win (Chase & Levine 2016; Lakeman-Fraser *et al*. 2016); a project focused on outreach and public engagement that generates meaningful scientific output.

## Acknowledgments

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# Chapter 5: Insect population trends and the IUCN Red List process

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## Abstract

Reliable assessment of extinction risk is a key factor in the preparation of Red Lists and in prioritizing biodiversity conservation. Temporal population trends can provide important evidence for such assessments, but imperfect sampling (observation errors) and short-term stochastic variation in population levels caused by environmental variability (process errors) can reduce the reliability of trends and lead to incorrect quantification of extinction risk. The assessment of insect taxa is likely to be particularly prone to these problems, due to the highly dynamic nature of many insect populations, driven by short life-cycles and sensitivity to environmental factors such as the weather. Using long-term United Kingdom monitoring data for 54 butterfly and 431 macro-moth species, we demonstrate the impact of insect population variability on the assessment of extinction risk using the International Union for Conservation of Nature (IUCN) Red List Criterion A (reduction in population size over the last 10 years). For both taxa, varying the start year of the 10-year population trend had a substantial effect on whether particular species met Red List thresholds and on the overall number of species assessed as threatened. We conclude that for these insect taxa strict application of the 10-year rule produces Red List classifications that are unacceptably biased by the start year. Use of long-term trends with adjustment based on species performance over the last decade may offer a pragmatic solution to this problem. We call for further IUCN guidance for practitioners undertaking Red List assessments of taxa with populations that have high temporal variability.

## Introduction

Biodiversity conservation practitioners rely on robust assessments of extinction risk (at global, regional, national and even local scales) to prioritise the use of limited resources. The Red List process developed by the International Union for Conservation of Nature (IUCN) plays an important role both as the global standard for extinction risk assessment (Miller *et al*. 2007; Mace *et al*. 2008) and, indirectly, in catalysing conservation activity. The Red List process itself is solely an objective, quantitative assessment of threat across taxa. Nevertheless, by providing a key input into prioritisation decisions made by practitioners and as a consequence of increased public and political support stemming from the credibility and reputation of the process, Red Lists are frequently a starting point for the development of conservation initiatives (Rodrigues *et al*. 2006; Hoffmann et al. 2008; Azam *et al*. 2016).

The IUCN process utilises criteria with quantitative thresholds based on population and distribution size and rate of decline in order to classify taxa into Red List threat categories (IUCN 2001). The development, application and misuse of these criteria have been documented (Akçakaya *et al*. 2006; Mace *et al*. 2008; Collen *et al*. 2016), as have the wider problems of applying them to insects and other invertebrates due to data constraints (e.g. Cardoso *et al*. 2011; van Swaay *et al*. 2011; Azam *et al*. 2016). Criterion A “Reduction in population size” depends solely on measures of population decline over a (potentially short) time-period of the most recent 10 years or three generations, whichever is longer, hereafter referred to as the “10-year rule” for simplicity. Thus widespread and common species, with large population sizes and ranges, can qualify as being threatened with extinction on Red Lists if they are undergoing rapid decline. Criterion A is justified because even large populations would eventually be driven to extinction by continuing decline (Mace *et al*. 2008), especially as other negative feedback loops may come into play at low population densities (e.g. Allee effects, genetic inbreeding), but also because the reduction in abundance of common and widespread species may be of particular significance to ecosystem structure and functioning (Gaston & Fuller 2008; Winfree *et al*. 2015).

However, the reliable measurement of species population trends that indicate extinction risk (and are being driven by anthropogenic processes such as habitat loss or climate change) for use in Red List Criterion A is made difficult by imperfect sampling (observation errors) and short-term stochastic variation in population levels caused by environmental variability (process errors) (Connors *et al*. 2014). Inaccurate detection of underlying species population trends can result in incorrect Red List classification (false positives i.e. incorrectly classifying a species as threatened and false negatives i.e. failing to classify a species that should be listed as threatened).

Investigations, using both empirical and simulated data, show that as process errors (and observation errors) increase, the reliable detection of population declines decreases across a range of different statistical techniques (Wilson *et al*. 2011; McCain *et al*. 2016). Trends assessed over short time periods, such as those required under the IUCN 10-year rule, are particularly sensitive to process errors, resulting in high levels of false positive and false negative species assessments (Connors *et al*. 2014; d'Eon-Eggertson *et al*. 2015). Concern has also been raised over the fundamental assumption that short-term declines are reliable predictors of ongoing decline (and, therefore, extinction risk) and authors have regularly advocated the use of long-term population data, where available, to improve the accuracy of extinction risk assessment (Dunn 2002; Porszt *et al*. 2012; Keith *et al*. 2015; White 2019). These findings undermine confidence in the classification of extinction risk using Criterion A in its current form (White 2019). However, these studies are based almost exclusively on vertebrate examples, where biological traits (e.g. generation times, population growth rates) and specific environmental drivers (e.g. human exploitation) may differ markedly from those of insects. Indeed, Connors *et al*. (2014) predict that the lowest rates of false-positive and false-negative classification errors under IUCN Red List Criterion A will occur for large-bodied, long-lived animal species.

These findings do not bode well for the application of Criterion A to insects. Most insect species have very short generation times (≤1 year), meaning that the 10-year rule is applied as a 10-year population trend over the most recent 10 years. By comparison, the same rule applied to long-lived vertebrates would see trends measured over longer time periods equating to three generations of the species concerned. In addition, the poikilothermic and ectothermic physiology of many insects results in climatic sensitivity that can drive large fluctuations in population size from generation to generation, particularly near altitudinal or latitudinal range margins (Oliver *et al*. 2014). Short-term climatic variation is a principal driver of inter-annual population change in UK butterflies and moths (Roy *et al*. 2001; Oliver *et al*. 2015; Palmer *et al*. 2017), alongside density dependence (Mills *et al*. 2017).

Due to anticipated high levels of process error (as well as potential observation error), 10-year population trends of insects may not be sufficiently reliable to enable the accurate classification of extinction risk in the Red List process, but rather reflect spurious responses to short-term environmental stochasticity. Thus, Red List classifications based on such trends are likely to be strongly affected by the start date of the 10-year trend, determined typically by factors such as policy development or funding availability that are unrelated to the population dynamics of the taxa being assessed.

The difficulty of detecting underlying declines from natural population fluctuations in short-term butterfly trends has been recognised previously (e.g. van Strien *et al*. 1997). Furthermore, the impact of temporal scale of trend measurement has been noted in comparisons of Red Lists produced using long-term versus 10-year trends (de Iongh & Bal 2007; van Swaay *et al*. 2011), and authors have recommended or developed adjustments to assessments under IUCN Criterion A to take long-term trends into consideration (Maes *et al*. 2012). In response, current IUCN guidance acknowledges this issue and sanctions optional use of data over a longer period to model population decline, especially for species with highly variable population levels, while still requiring trends to be measured over the most recent 10 years (IUCN 2017). Nevertheless, the 10-year rule remains fundamental to IUCN Criterion A and practitioners can continue to assess the threat levels of short-lived species based on just a decade of population data.

The purpose of this study is to highlight, from a practitioner's perspective, problems with the application of 10-year population trends in the Red Listing of insects and to seek further advice from IUCN. Specifically, we quantify, for the first time, the direct implications of high levels of inter-annual population variation (process errors) on Red List classification under IUCN Criterion A. Our assessment of two United Kingdom (UK) insect taxa for which standardised population monitoring data are available, butterflies and macro-moths, is then used to consider whether the 10-year rule is appropriate for such taxa.

## Methods and results

We consider two case studies using UK insect population data for butterflies and macro-moths derived from long-term (40+ years) monitoring schemes to assess the impact of species' population variability on Red Listing using IUCN Criterion A. Although butterflies and moths are closely related taxa in the Order Lepidoptera, considering them as separate case studies is appropriate and informative as the monitoring schemes and datasets for each are independent and utilise different methodologies (fixed-width line counts for butterflies and point counts using light-traps for macro-moths) to sample diurnal and nocturnal insect communities respectively. In addition, long-running time series of standardised abundance for insect taxa are rare in the UK and globally; the only other insect taxon for which data are available over a comparable duration in the UK are aphids (Order Hemiptera, Superfamily Aphidoidea), although the geographical coverage is much less extensive (Thomas 2005).

We also varied two aspects of the assessment method in each case study: standardisation of time periods across species and the use of population trends with or without statistical significance. First, in the butterfly case study, the 10-year periods being compared were standardised (i.e. they started in the same year for each species), whereas in the macro-moth case study, the 10-year time periods varied between species according to data availability. Second, butterfly population trends were assessed against the IUCN threat category thresholds irrespective of whether the trends were statistically significant, while in the macro-moth study only statistically significant population trends were used in the assessment. These alternatives were used to represent the range of different approaches likely to be employed by practitioners undertaking Red List assessment depending on the form and availability of data.

It should be noted that the case studies do not represent the application of a complete Red List process, but are indicative assessments of the potential impact of one IUCN criterion on the outcome. A full Red List procedure would utilise other criteria based on geographical range and population size (depending on data availability) and also, when carried out at a regional level, an important additional consideration is the potential for the extinction risk of a taxon to be influenced by movement of individuals into or out of the region being assessed (IUCN 2012). However, our consideration of Criterion A in isolation is relevant because threatened Red List status is conferred under the precautionary principle - as long as a taxon meets the threshold for a single criterion then it can be classified as threatened. Thus, false positive assessments under Criterion A (or any criterion) could exert substantial influence over completed Red Lists.

*Case study 1: UK butterflies*

We considered the impact of arbitrary start date on the Red List outcomes for UK butterflies based on published 10-year population trends derived by linear regression from the UK Butterfly Monitoring Scheme (UKBMS) over six consecutive years (www.ukbms.org; Pollard & Yates 1993; Roy *et al*. 2015). The standardised, annual monitoring of butterfly abundance by the UKBMS at over 1,000 sites generates robust population data used by the Government to assess biodiversity trends (Brereton *et al*. 2011; Eaton *et al*. 2015). Despite low observation error, 10-year UKBMS population trends for many species fluctuate considerably from year to year, reflecting stochastic environmental variation (process error) (Table A5.1). The 10-year population trends for each species were assessed against the IUCN Criterion A2 thresholds (A2 being for population trends where the reduction or its causes may not have ceased or may not be understood or may not be reversible) and species allocated to threat categories accordingly. Trend values were utilised in the assessment irrespective of their statistical significance (in contrast to the macro-moth case study). Thus, six classifications were produced using population trends for six 10-year periods, each starting one year after the previous one (i.e. 2001-2010, 2002-2011, 2003-2012, 2004-2013, 2005-2014 and 2006-2015). In addition, the median, lower and upper quartile population trends were calculated for each species from the six 10-year trend values and these were also assessed against IUCN Criterion A2.

Fifty-four species (of the 59 resident or common migrant butterfly species present in the UK) had UKBMS 10-year population trends for all six periods considered. There was considerable variation in the total number of species qualifying for Red List categories between classifications and for individual species across classifications. An average of 18.5 species met the IUCN

Criterion A2 thresholds for threatened status (i.e. Critically Endangered ≥80% population decrease, Endangered ≥50% decrease or Vulnerable ≥30% decrease) per period, but the number of species qualifying ranged from 13 (24% of species) to 29 (54% of species) (Table 5.1). Twenty species (37% of the total) were consistently classified across the six different time periods (i.e. they either always (3 species) or never (17 species) qualified as threatened), but 34 species (63%) qualified as threatened in some periods and not others (Table 5.2). Removing the three common migratory species from the sample had no qualitative effect on the overall pattern.

Using the median population change value over the six 10-year periods for the Red List assessment produced 18 threatened species, the lower quartile trend value led to 25 threatened species and the upper quartile trend just 8 species (Table 5.1). The threat category assigned to a particular species frequently differed between the median, lower and upper quartile values (Table 5.2).

**Table 5.1** Number of UK butterfly species (of 54 species assessed) meeting Red List threat thresholds under IUCN Criterion A2 (reduction in population size) on basis of 10-year UKBMS population trends over different year ranges and the median, lower and upper quartile trend values across the periods. CR = Critically Endangered (decrease ≥80%), EN = Endangered (decrease ≥50%), VU = Vulnerable (decrease ≥30%). These classifications do not represent the final outcomes of a full Red List process.

| | **2001-2010** | **2002-2011** | **2003-2012** | **2004-2013** | **2005-2014** | **2006-2015** | **Median** | **Lower Qrt** | **Upper Qrt** |
|---|---|---|---|---|---|---|---|---|---|
| **CR** | 2 | 4 | 5 | 5 | 2 | 2 | 2 | 6 | 1 |
| **EN** | 10 | 8 | 14 | 11 | 7 | 4 | 11 | 12 | 4 |
| **VU** | 6 | 3 | 10 | 4 | 7 | 7 | 5 | 7 | 3 |
| **Total** | **18** | **15** | **29** | **20** | **16** | **13** | **18** | **25** | **8** |

**Table 5.2** Red List threat thresholds met by UK butterflies under IUCN Criterion A2 (reduction in population size) on basis of 10-year UKBMS population trends over different year ranges, and the median (MT), lower (LQ) and upper quartile (UQ) values of these trends. CR = Critically Endangered (decrease ≥80%), EN = Endangered (decrease ≥50%), VU = Vulnerable (decrease ≥30%). Empty cells indicate that the species did not qualify as threatened. Species are ranked by the total number of classifications in which they reach Red List thresholds. These classifications do not represent the final outcomes of a full Red List process. * Common migrant species in the UK.

| **Taxon** | **2001 - 10** | **2002 - 11** | **2003 - 12** | **2004 - 13** | **2005 - 14** | **2006 - 15** | **Total** | **MT** | **LQ** | **UQ** |
|---|---|---|---|---|---|---|---|---|---|---|
| *Colias croceus** | EN | CR | CR | CR | EN | VU | 6 | EN | CR | EN |
| *Melitaea athalia* | EN | EN | EN | CR | EN | CR | 6 | EN | CR | EN |
| *Vanessa cardui** | VU | CR | CR | CR | CR | CR | 6 | CR | CR | CR |
| *Pyronia tithonus* | VU | VU | EN | EN | VU |  | 5 | VU | EN | VU |
| *Thecla betulae* | EN | EN | EN | EN | EN |  | 5 | EN | EN | EN |
| *Thymelicus lineola* | CR | CR | CR | CR | EN |  | 5 | CR | CR | EN |
| *Argynnis adippe* | EN | EN | CR | EN |  |  | 4 | EN | EN |  |
| *Euphydryas aurinia* |  |  | EN | EN | EN | EN | 4 | EN | EN |  |
| *Lasiommata megera* | EN | EN | EN | EN |  |  | 4 | EN | EN | VU |
| *Leptidea sinapis* | EN | EN | EN | EN |  |  | 4 | EN | EN |  |
| *Limenitis camilla* |  |  | VU | EN | VU | VU | 4 | VU | VU |  |
| *Satyrium pruni* |  |  | VU | CR | CR | EN | 4 | EN | CR |  |
| *Satyrium w-album* |  |  | EN | EN | EN | EN | 4 | EN | EN | VU |
| *Thymelicus acteon* | CR | CR | CR | EN |  |  | 4 | EN | CR |  |
| *Thymelicus sylvestris* | EN | EN | EN | VU |  |  | 4 | VU | EN |  |
| *Aglais urticae* | EN | EN | EN |  |  |  | 3 | EN | EN |  |
| *Apatura iris* |  |  | EN |  | VU | VU | 3 |  | VU |  |
| *Celastrina argiolus* |  |  |  | EN | EN | VU | 3 | VU | EN |  |
| *Coenonympha tullia* |  |  |  | EN | VU | EN | 3 |  | EN |  |
| *Hamearis lucina* | EN | EN | EN |  |  |  | 3 |  | EN |  |
| *Polyommatus bellargus* |  |  | VU |  | VU | VU | 3 | VU | VU |  |
| *Aricia agestis* |  |  | VU | VU |  |  | 2 |  |  |  |
| *Callophrys rubi* |  |  |  | VU | VU |  | 2 |  | VU |  |
| *Erebia aethiops* | EN | VU |  |  |  |  | 2 |  | VU |  |
| *Gonepteryx rhamni* |  |  | VU | VU |  |  | 2 |  |  |  |

| | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|
| *Hesperia comma* | VU | | EN | | | | 2 | | VU | |
| *Maniola jurtina* | | VU | VU | | | | 2 | | | |
| *Plebejus argus* | VU | | VU | | | | 2 | | | |
| *Vanessa atalanta** | | | | | VU | VU | 2 | | VU | |
| *Aglais io* | | | VU | | | | 1 | | | |
| *Aricia artaxerxes* | VU | | | | | | 1 | | | |
| *Boloria euphrosyne* | VU | | | | | | 1 | | | |
| *Lycaena phlaeas* | | | | | | VU | 1 | | | |
| *Pieris brassicae* | | | VU | | | | 1 | | | |
| *Pieris rapae* | | | EN | | | | 1 | | | |
| *Polygonia c-album* | | | VU | | | | 1 | | | |
| *Polyommatus icarus* | | | EN | | | | 1 | | | |
| *Anthocharis cardamines* | | | | | | | 0 | | | |
| *Aphantopus hyperantus* | | | | | | | 0 | | | |
| *Argynnis aglaja* | | | | | | | 0 | | | |
| *Argynnis paphia* | | | | | | | 0 | | | |
| *Boloria selene* | | | | | | | 0 | | | |
| *Coenonympha pamphilus* | | | | | | | 0 | | | |
| *Cupido minimus* | | | | | | | 0 | | | |
| *Erynnis tages* | | | | | | | 0 | | | |
| *Favonius quercus* | | | | | | | 0 | | | |
| *Hipparchia semele* | | | | | | | 0 | | | |
| *Maculinea arion* | | | | | | | 0 | | | |
| *Melanargia galathea* | | | | | | | 0 | | | |
| *Ochlodes sylvanus* | | | | | | | 0 | | | |
| *Pararge aegeria* | | | | | | | 0 | | | |
| *Pieris napi* | | | | | | | 0 | | | |
| *Polyommatus coridon* | | | | | | | 0 | | | |
| *Pyrgus malvae* | | | | | | | 0 | | | |

### *Case study 2: UK macro-moths*

A second case study, using population data for 431 UK macro-moths (hereafter “moths”), was undertaken to assess the wider applicability of the results for butterflies.

Monitoring of adult moth numbers has been carried out across the UK since 1968, as part of the Rothamsted Insect Survey (RIS) run by Rothamsted Research (www.rothamsted.ac.uk/insect-survey). Standardised light-traps operate at approximately 80-100 sites annually, on every night of the year and all moths attracted into the traps are retained for identification by professional staff or expert volunteers (Conrad *et al*. 2004). The data have been used to assess long-term change in moth biodiversity, including as part of official Government indicators (Conrad *et al*. 2006; Eaton *et al*. 2015; Burns *et al*. 2018).

For this case study, we present a preliminary analysis of RIS abundance data using the Generalized Abundance Index (GAI) approach (Dennis *et al*. 2016). RIS count data were extracted for UK resident moth species in the families Hepialidae, Cossidae, Sesiidae, Limacodidae, Zygaenidae, Drepanidae, Lasiocampidae, Endromidae, Saturniidae, Sphingidae, Geometridae, Notodontidae, Erebidae, Noctuidae and Nolidae (Agassiz, Beavan & Heckford 2013). Species that occur in the UK only as immigrants were excluded. Daily species count data for the full RIS time series (1960-2015) were analysed using the GAI method and trends assessed using linear regression. The data were not filtered prior to analysis, but *post hoc* tests on the GAI for the entire time series of data for each species were used to identify statistically unreliable models. Species were excluded from the case study where the results contained indices for less than 10 years and/or where the number of years with missing indices was greater than 30% of the total series. In addition, the annual index values and their standard errors were assessed and species exhibiting extreme indices (indices <zero or >4) or exceptionally large standard errors (standard error >1) were also excluded from the case study.

Having excluded species that did not meet the minimum statistical requirements, a series of five, overlapping 10-year population trends were calculated for the remaining species. First, in keeping with the IUCN 10-year rule, GAI values for the most recent 10 years available for each moth species were used to derive a population trend by fitting linear regressions. The key parameters (e.g. slope, intercept, statistical significance) of each of these linear models were stored, and measures of annual growth rate and proportional change over the 10-year time period were calculated from these parameters. For the majority of species the most recent 10-year period was 2006-2015. However, as data availability varied from species to species, e.g. because rapidly declining species become so scarce that they are no longer caught at all in the RIS monitoring network, the start/end year of this most recent 10-year period was not the same for all species.

Next, this process was repeated four times for every moth species, on each occasion starting the 10-year period one year earlier. Only the GAI values for each 10-year period were used to calculate the population trend in each instance. This resulted in five 10-year population trends per species, each trend lagged by one year: t (the most recent 10 years), t-1, t-2, t-3 and t-4. For the majority of species, the five trends covered the periods 2006-2015, 2005-2014, 2004-2013, 2003-2012 and 2002-2011, but some extended back into the 1990s and, in one case, the 1980s. For each time period, 10-year species population trends that were statistically significant at $p < 0.05$ were then assessed against IUCN Criterion A2 thresholds to provide a threat (extinction risk) classification.

Population trends for a total of 431 moth species, which had statistically reliable long-term GAI models, were assessed across five overlapping 10-year time periods (Table A5.2). 109 species (25% of the total) had statistically significant 10-year population trends that met IUCN Criterion A2 thresholds for Red List threat categories (i.e. Critically Endangered ≥80% population decrease, Endangered ≥50% decrease or Vulnerable ≥30% decrease) in at least one of the five time periods. The remaining 322 species (75% of the total) did not meet these conditions in any of the five 10-year periods. However, of the 109 species that qualified as threatened, only five (4.6%) did so in all five of the time periods; the remaining 104 moth species were variable, qualifying for the Red List in some time periods but not in others, despite the fact that the five time periods were offset by only one year in each case.

The number of moths qualifying under Criterion A2 varied considerably between the time periods (Table 5.3). Most dramatically, the difference of a single year between period t-3 and t-4 reduced the number of qualifying species from 62 (14% of the total number of species assessed) to just 20 (5%).

While appropriate for use in this case study and for demonstrating the variation in trend magnitude from year to year, it should be noted that this is a preliminary analysis of RIS data and the proportional change values over time for individual species may differ when a more detailed analysis is carried out.

**Table 5.3** Number of UK macro-moth species (of 431 species assessed) meeting Red List threat thresholds under IUCN Criterion A2 (reduction in population size) on the basis of preliminary 10-year RIS population trends representing the most recent 10-year period (t) and preceding 10-year periods each starting one year earlier than the previous (t-1, t-2, t-3, t-4). These classifications do not represent the final outcomes of a full Red List process.

| | **t** | **t-1** | **t-2** | **t-3** | **t-4** |
|---|---|---|---|---|---|
| Critically Endangered | 14 | 17 | 13 | 13 | 4 |
| Endangered | 27 | 24 | 35 | 37 | 11 |
| Vulnerable | 5 | 5 | 9 | 12 | 5 |
| **Total** | **46** | **46** | **57** | **62** | **20** |

### Discussion

The case studies using UK butterfly and macro-moth population time series revealed large discrepancies between Criterion A Red List classifications produced using trends that differed by just a single year. For individual species, the temporal patterns of Red List qualification might reflect genuinely improving or deteriorating levels of extinction risk. On the other hand, and as indicated by the dynamic nature of many species population trends between years (Table A5.1, Table A5.2), patterns may be artefacts of process errors driven by environmental (particularly climatic) variability. Whatever the specific cause of the intra-species variation, the application of the 10-year rule, and specifically the requirement for the population trend to be measured over the most recent 10 years, leads, in our opinion, to a scientifically unacceptable dependency of the Red List classification outcome on the year in which the process is undertaken.

The use and misuse of IUCN Red List criteria has been considered frequently in the literature (Eaton *et al*. 2005; Akçakaya *et al*. 2006; Collen *et al*. 2016), but the specific issue concerning the use of short-term (the most recent 10 years or three generations) population trends to classify insect taxa under Criterion A has not been addressed. Many insect species naturally undergo highly variable and erratic population dynamics, due to environmental variation (Williams 1961) or density-dependence effects (Hanski 1990), and, as illustrated in the case studies using UK butterflies and moths, this may impact significantly on the Red List classifications.

Studies using vertebrate population data have concluded that longer time series can improve the assessment of extinction risk under Criterion A, and practitioners undertaking Red Listing of butterflies have highlighted the same issue (van Swaay *et al*. 2011; Maes *et al*. 2012). Indeed the current IUCN guidelines reflect this, suggesting that using data from a longer time period to fit a statistical model of population decline may be preferable for species that have widely fluctuating or oscillating population dynamics (in Section 4.5.1, IUCN 2017). Nevertheless, the IUCN guidelines go on to stress that having fitted the model, the proportional decline should still be calculated over the most recent 10 years or three generations, as per the 10-year rule. In light of our results, we do not consider this guidance to be sufficient. It is optional, dependent on the availability of long-term data and relies on practitioners being familiar with the detailed IUCN guidance. Even if applied, the requirement to calculate change over the most recent 10 years is unlikely to ameliorate the problem illustrated by our case studies, as high levels of inter-annual population variability within the 10-year period are still likely to strongly skew trends and therefore Red List assessments. More fundamentally, the 10-year rule remains the basis of IUCN Criterion A and can be used to determine the extinction risk of species without use of longer-term data. Our results, quantifying the impacts of 10-year trend start year on the number and identity of species meeting Red List thresholds, suggest that this is inappropriate for UK butterflies and macro-moths and potentially for many other insect and invertebrate taxa around the world.

An obvious solution to the problem is to measure population trends over a longer period of time rather than the last 10 years. Linear trends over the 40+ year time series available for both UK butterflies and moths dampen the effects of annual variation, providing a more robust assessment of population change.

From such long-term population trends, annual rates of change can be used to calculate a 10-year trend for each species that can be assessed against the IUCN Criterion A thresholds. The important distinction is that this is a population change measured over an average 10-year period of a longer time interval, rather than being measured over the most recent 10 years.

This approach brings other benefits too. IUCN guidance explicitly warns against interpreting the downward phase of population cycles as a reduction under Criterion A (Section 4.5, IUCN 2017). While there are well-established cases of population periodicity in moths (e.g. Berryman 1996; Johnson *et al*. 2006; Bell *et al*. 2012), for most species it is unclear whether populations are truly cyclical or simply erratic, making it difficult to apply the IUCN guidance. Trends derived from a long-term time series will be less prone to misinterpretation and misclassification caused by unrecognised population cyclicity. In addition, if practitioners determine that only statistically significant population trends should be used to assign species to Red List threat categories, trends calculated over just the most recent 10 years are unlikely to attain significance, when populations are naturally variable, even if reductions (or increases) are very large, as a result of the small number of data points.

Despite all of these benefits, the key problem with adopting such an approach for Red Listing is that the IUCN guidelines are clear that Criterion A should represent the recent population trend of a taxon and not take account of historical declines. Indeed the 10-year rule is specifically there to ensure that species that have undergone major declines in the past, but are currently stable or recovering, are not classified as threatened under the IUCN Red List process (unless there is sufficient evidence of future threats to support a projected decline that meets threshold levels) (Section 5.4 and 5.5, IUCN 2017).

As a compromise, which avoids the spurious variability of the 10-year rule yet embraces the spirit of reflecting recent population decline, we propose a two-step process to the implementation of IUCN Criterion A with insect population trends. First, long-term data are used to derive an average 10-year trend over the full time series, which is then assessed against the IUCN quantitative thresholds to produce a provisional threat classification for each species.

Second, a population trend derived just from the last 10 years of data is calculated for each species and used to adjust the provisional threat classification using expert judgement. Thus, the threat status of species with a long-term population decline but recent stability or recovery would be downgraded, while that of species with both long-term and recent declines would be maintained, or even be increased if the recent trend shows an increasing rate of population reduction. As with all elements of the Red Listing process, it would be essential to document the basis for upgrading or downgrading the threat category of each species to ensure transparency.

An alternative approach to adjusting the classification produced by Criterion A was implemented by Maes *et al*. (2012) when applying the IUCN criteria to butterflies in Flanders (Belgium). They calculated 10-year rates of change (from occurrence rather than abundance data) and applied the IUCN Criterion A thresholds to produce an initial classification for each species. They then upgraded species by one Red List category if they had shown >50% historical distribution decline over a longer time period (c.30 years).

The IUCN Red List process is an important force in biodiversity conservation and has been successfully applied at global, regional and national levels to a wide range of taxa, including insects and other invertebrates (Collen *et al*. 2012). This has been aided in recent years by the development of new statistical techniques to extract reliable trends from species occurrence data (Isaac *et al*. 2014; Maes *et al*. 2015; Dennis *et al*. 2017). Long- and short-term temporal trends can now be derived from annual indices generated by occupancy modelling (Burns *et al*. 2018) and could be used with Criterion A to facilitate Red List assessment of many more invertebrate taxa in many more countries and regions. To our knowledge, the sensitivity of occupancy trends to inter-annual variability has not been examined and this should be a focus of further research prior to the use of such trends under the 10-year rule in Red List assessment for insects.

Robust population monitoring remains the gold standard, however, for measuring biodiversity change (Roy *et al*. 2007; Morecroft *et al*. 2009) and the geographical and taxonomic extent of such schemes for insects continues to expand (van Swaay *et al*. 2008; Carvell *et al*. 2018; Matechou *et al*. 2018). Where available, population monitoring data should be utilised in Red List assessments, yet the natural variability of insect populations presents a dilemma for conservation practitioners in applying the Red List ‘10-year rule’. Given the variability illustrated here with case studies on UK butterfly and macro-moth populations, practitioners should be extremely wary of assigning extinction risk to insects based on only the last 10 years of population data as per IUCN Criterion A. Our examples suggest that longer time series of data are required to produce a robust assessment, but trends measured over a long time period are likely to be less indicative of the current extinction risk of a species. While we have proposed one possible compromise solution to ameliorate this issue, and other practitioners may adopt other approaches, further consideration and advice from IUCN on the application of Criterion A for species with high process errors would be very welcome.

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# Chapter 6: Opinions of citizen scientists on open access to UK butterfly and moth occurrence data

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## Abstract

Citizen science plays an increasingly important role in biodiversity research and conservation, enabling large volumes of data to be gathered across extensive spatial scales in a cost-effective manner. Open access increases the utility of such data, informing land-use decisions that may affect species persistence, enhancing transparency and encouraging proliferation of research applications. However, open access provision of recent, fine-scale spatial information on the locations of species may also prompt legitimate concerns among contributors regarding possible unintended negative conservation impacts, violations of privacy and commercial exploitation of volunteer-gathered data. Here we canvas the attitudes towards open access of contributors (104 regional co-ordinators and 510 recorders) of species occurrence records to two of the largest citizen science biodiversity recording schemes, the UK's Butterflies for the New Millennium project and National Moth Recording Scheme. We find that while the majority of participants expressed support for open access in principle, most were more cautious in practice, preferring to limit the spatial resolution of records, particularly of threatened species, and restrict commercial reuse of data. In addition, citizen scientists' opinions differed between UK countries, taxonomic groups and the level of involvement volunteers had in the schemes. In order to maintain successful and democratic citizen science schemes, organisers, funders and data users must understand and respect participants' expectations and aspirations regarding open data while seeking to optimise data use for scientific and societal benefits.

## Introduction

There is growing expectation and demand for open access to data in many areas of public life including science. In addition to the accepted scientific requirements of transparency and reproducibility, and the responsibility of public funding, this demand has been driven by the development of “big data” technologies enabling the storage and analysis of huge quantities of information (Arzberger *et al*. 2004; Farley *et al*. 2018). Scientists are increasingly willing to share data publicly (Tenopir *et al*. 2015), enabling other researchers to utilise and build upon freely-available archived data, resulting in benefits for society. An open access culture has developed in some scientific fields, notably genetics and genomics (Noor *et al*. 2006), although even here ethical concerns remain (McGuire *et al*. 2011; McEwen *et al*. 2013; Choudhury *et al*. 2014).

Ecologists, however, have been relatively slow to embrace open data, despite its potential to address many urgent, global, environmental pressures (Hampton *et al*. 2013; Poisot *et al*. 2013; Kenall *et al*. 2014; Soranno *et al*. 2015). Progress towards a more open approach in ecology is hindered by technological and cultural barriers, but solutions and incentives have emerged, alongside new obligations for public data archiving from funding organisations and scientific journals (Reichman *et al*. 2011; Michener 2015; Nosek *et al*. 2015; Culina *et al*. 2018a). Nevertheless, concerns remain about open access to ecological data, and while the views of scientists and organisations have been reported (Moles *et al*. 2013; Mills *et al*. 2015; Pearce-Higgins *et al*. 2018; Tulloch *et al*. 2018), the opinions of citizen scientists themselves have been overlooked.

Ecological data gathered through citizen science projects are increasingly useful, particularly for biodiversity monitoring and conservation (Chandler *et al*. 2017; Sullivan *et al*. 2017; Soroye *et al*. 2018). Unrestricted access to and reuse of citizen science ecological data maximises the societal and scientific returns on the efforts of volunteers; for example disclosure of locations of threatened species can encourage informed decision making about land-use changes that might impact biodiversity, improve species’ trend assessments, facilitate applied scientific research and help engage landowners, funders, politicians and the public in conservation (Tulloch *et al*. 2018). However, in the context of open access, citizen science data differ fundamentally from those collected in professional scientific research because the data are contributed by volunteers, who have their own views on data accessibility. It is widely expected that citizen science ecological data will be open access (Groom *et al*. 2017; Robinson *et al*.

2018), perhaps because it is supposed that people who contribute willingly and without material reward to citizen science projects would assume, or even insist, that their data are freely shared and publicly accessible. This assumption may not be justified, in part because the large number of citizen scientists are bound to encompass a diversity of views but also, specifically, because some participants have been engaged in gathering ecological data under different data exchange principles long before the advent of the “big data” era and the contemporary pressure for open access. Indeed, while the term citizen science was coined in the mid-1990s and the field has burgeoned since then (Silvertown 2009; Pocock *et al*. 2017), there is a long tradition of amateur naturalists gathering ecological, and particularly biogeographical, information (Miller-Rushing *et al*. 2012; Pocock *et al*. 2015; Strasser *et al*. 2019). In this tradition, the individual’s motivation to observe and study nature may have little to do with science or biodiversity conservation, leading to mismatches and tensions between the expectations of the scientific establishment and these participants in projects that are nowadays labelled as ‘citizen science’ (Ellis & Waterton 2004).

Thus, while some citizen science projects have an explicitly open data ethos (e.g. eBird, Sullivan *et al*. 2014), others do not (Groom *et al*. 2017). This may simply be because projects and their participants are continuing the historical legacy of mindsets, relationships and practices formed long before the advent of modern citizen science (Strasser *et al*. 2019) and do not conform to its expectations around open access. Alternatively, access to data may be restricted deliberately due to legitimate concerns from project organisers (Pearce-Higgins *et al*. 2018; Tulloch *et al*. 2018). One such concern is that unintended negative consequences of open access, for example harm to threatened species, could lead citizen scientists to cease participation, undermining project viability. It is important, therefore, that organisers, funders and users of citizen science are mindful of the views of participants regarding open access. While the motivations of citizen scientists taking part in biodiversity projects have been surveyed (Evans *et al*. 2005; Hobbs & White 2012; Wright *et al*. 2015; Domroese & Johnson 2017), their attitudes towards the onward use of the data that they contribute, and on the specific issue of open access to data, have rarely been considered (Ganzevoort *et al*. 2017).

These issues are of interest and importance to governmental and non-governmental organisations involved in conservation and research. For example, the charity Butterfly Conservation runs long-term citizen science schemes focussed on butterflies and moths (Lepidoptera) in the United Kingdom (UK). The schemes rely upon collaboration between paid staff (organising and promoting the schemes, managing databases, undertaking research and providing feedback to participants) and unpaid volunteers (undertaking species recording, computerisation and verification of records). Tens of thousands of volunteers are involved annually and the schemes have generated datasets that underpin assessments of UK Lepidoptera biodiversity change (e.g. Fox *et al*. 2014; Fox *et al*. 2015) and the delivery of species conservation (Ellis *et al*. 2012), as well as research e.g. into the impacts of environmental drivers such as climate change (e.g. Mason *et al*. 2015; Martay *et al*. 2017; Pearce-Higgins *et al*. 2017). In most cases, the data assembled through these schemes are not currently open access. Yet, given the considerable potential benefits for both biodiversity protection and scientific research of increasing access to these data, as well as the ethical impetus towards greater inclusivity (Soranno *et al*. 2015), the availability of these datasets should be reviewed and weighed against possible negative repercussions (e.g. impacts on threatened species or habitats, intrusion on participants' privacy or damage to partnerships with private landowners who have allowed access to otherwise closed land).

Therefore, to inform such a review and to provide practical recommendations to designers, organisers and funders of similar citizen science projects, we conducted surveys of volunteer participants in Butterfly Conservation recording schemes to seek a nuanced understanding of their views on open access to butterfly and moth occurrence data. Our study extends the approach of Ganzevoort *et al*. (2017), the only similar survey that we are aware of, by exploring the influence of spatial resolution, deferred data release and species threat on the attitudes of two different groups of volunteers with differing roles and levels of involvement in citizen science schemes, as well as contrasting the opinions of recorders of different taxa and in different UK countries. Our principal aim was to document the attitudes of these different groups of participants and understand how these may influence transition towards more open models of data accessibility. We did not seek to explore the motivations or values underlying participants' attitudes to open access and acknowledge that, as a result, the findings in this respect are limited. However, in addition to quantifying opinions, we sought to test the following hypotheses: 1) if the main concerns of citizen scientists related to potential damage to butterflies, moths or their habitats, rather than about privacy, confidentiality or intellectual property rights, then they would be more reluctant to allow open access to records of threatened species compared to widespread ones and 2) that unwillingness to make threatened species records open access would be ameliorated by limiting (blurring) spatial location information and postponing the release of records for long periods (five or more years).

## Methods

### *Focal citizen science projects*

The opinions towards open access of contributors to two UK-wide citizen science projects organised by Butterfly Conservation, Butterflies for the New Millennium (BNM; Asher *et al*. 2001) and the National Moth Recording Scheme (NMRS; Fox *et al*. 2011), were ascertained by questionnaires. The BNM was launched in 1995 and has, to date, collated 12.7 million butterfly species occurrence records covering the period 1690-2017. The NMRS commenced in 2006, initially focusing on macro-moth occurrence records (although it has now been extended to include all moth species), and has compiled 25 million macro-moth records for the period 1746-2016. These projects are among the largest citizen science biodiversity monitoring schemes globally, but the majority of BNM and NMRS records are not currently open access.

The flow of species occurrence records through the BNM and NMRS projects is organised in the same way. Observations made by citizen scientist recorders are sent to regional co-ordinators (also known as County Recorders), who are expert volunteers with the responsibility to collate and verify sightings for their area and maintain a local dataset of records. Copies of these local datasets are then pooled annually and, following further checks, added to the BNM or NMRS databases. At the time of this study, the BNM project included 65 regional co-ordinators and the NMRS 94. A few individuals fulfilled both roles for their area. The total numbers of citizen scientist recorders participating in the BNM and

NMRS annually is unknown, because of inconsistencies in the way that individual recorder identities are logged across the schemes. However, given that each scheme currently collates c.1 million new records per annum, it is likely that there are tens of thousands of contributors at present. Some recorders take part in one but not the other scheme, whereas others contribute sightings to both.

The BNM and NMRS schemes collate opportunistic sightings of species from any location in the UK and on any date. Although there are minimum information standards for valid sightings, there are no sampling protocols – participants can record where, when and for as long as they wish. This traditional model of natural history recording (Pocock *et al*. 2015), separates the schemes on the one hand from systematic monitoring programmes with rigorous sampling protocols undertaken by experienced amateur or professional naturalists (e.g. the UK Butterfly Monitoring Scheme, Brereton *et al*. 2011; North American Breeding Bird Survey, Sauer *et al*. 2013) and, on the other, from modern citizen science projects that often aim to engage people with no previous involvement (e.g. Big Butterfly Count, Dennis *et al*. 2017; Great Pollinator Project, Domroese & Johnson 2017). Thus, while all BNM and NMRS participants are volunteers, their natural history expertise and recording behaviour vary greatly, as has been found in other biodiversity surveillance projects (Boakes *et al*. 2016; Everett & Geoghegan 2016).

Although it is difficult to categorize BNM and NMRS recorders on the basis of levels of engagement or expertise, different volunteer roles within the schemes provide a clear dichotomy; individual regional co-ordinators are essential to the functioning of the schemes in a way that individual recorders are not, as without a regional co-ordinator in place no new records for that area will be provided to the scheme. While the opinions of both groups are important, the integral role of regional co-ordinators in the operation of the schemes necessitates an understanding of their attitudes to data sharing of the records in their custodianship as part of any prospective shift toward open access to the BNM and NMRS data. In addition, as curators of local datasets of species occurrence records, regional co-ordinators are likely to be familiar with the pros and cons of open access and, as expert naturalists, their views will be shaped by the traditions of data exchange within amateur natural history (Ellis & Waterton 2005; Ellis *et al*. 2005).

### *Questionnaires*

Separate questionnaires were designed to elucidate the views of regional co-ordinators and recorders and surveys were undertaken in May and June 2017. A longer questionnaire was used for regional co-ordinators so that we could gain a detailed understanding of the views of this key group of volunteers, while a much shorter, ‘light touch’ and entirely anonymous questionnaire was developed for recorders to maximise participation in the study.

### *Regional co-ordinator questionnaire*

The questionnaire for regional co-ordinators (Appendix 6) aimed to ascertain the current level of support for and against open access and to gauge how such attitudes vary between volunteers in schemes for different taxa, in different countries and in response to perceived risk of negative impacts. Even when data are made publicly accessible, potential risks to species, habitats, sites or citizens can be moderated by restricting the information that is made available, by delaying the release of data and by legally restricting the uses to which data can be put. Thus, general support for open access was assessed by responses on a 10-point numerical scale (from 1 = serious reservations to 10 = strongly in favour), but subsequent questions asked participants to consider the appropriate spatial resolution of open records (i.e. how much records are blurred to conceal the precise location of species occurrence, with options of full capture resolution or blurring to 1km x 1km square, 2km x 2km square or 10km x 10km square), whether there should be a time lag before records are made public (with options of no lag, 5 year, 10 year or 20 year lags) and on the type of Creative Commons license that should be applied to open access UK butterfly and moth data. Developed as an alternative to traditional ‘all rights reserved’ copyright, Creative Commons licenses enable the copyright holder to choose which rights to reserve and which to waive, and have been widely adopted in many fields of human endeavour, including biodiversity monitoring (Hagedorn *et al*. 2011; Groom *et al*. 2017). Regional co-ordinators were asked for their opinion on the most appropriate of three Creative Commons licenses for UK butterfly and moth occurrence data; Zero (CC0), which has no restrictions on reuse, Attribution (CC-BY), which requires users to acknowledge the author/source, and Attribution-NonCommercial (CC-BY-NC), which requires acknowledgement and restricts reuse to non-commercial applications.

In addition to controlling data availability and use, the rarity or threat levels of taxa are likely to influence the perception of risk stemming from open access. The questionnaire sought to quantify this by asking respondents to consider the appropriate spatial resolution for open access records separately for widespread and threatened species. ‘Widespread’ and ‘threatened’ were not defined, so respondents used their own interpretation. In addition, regional co-ordinators were asked whether there were taxa or specific populations of taxa in their area that would require a more restrictive approach than the various open access options already discussed.

In total, the regional co-ordinator questionnaire included six questions with multiple-choice or scaled answers. Respondents were asked to provide their name and the geographical area for which they fulfil the role of regional co-ordinator. Questions were not obligatory and not all respondents completed all questions.

The questionnaire was sent by email attachment as a Microsoft Word document with a covering letter (Appendix 6) to all UK regional co-ordinators in the BNM and NMRS networks on 10 May 2017. Regional co-ordinators were given until the end of May 2017 to respond, although responses received by 7 June 2017 were included in the analysis.

*Recorder questionnaire*

A simpler questionnaire (Appendix 6) was designed to canvas recorders’ views on open access and how recording behaviour might change in response to it. Just four multiple-choice questions were asked; two to segment respondents by UK country and taxonomic interest (recording butterflies, moths or both) and two relating to open access. Recorders were asked for their preferred open access spatial resolution for their own records via the BNM and NMRS schemes. Three options were provided: all records open at full capture resolution (i.e. the same level of spatial resolution as submitted by the recorder), widespread species at full resolution but scarce/threatened species at a summary (i.e. blurred) resolution, and all records at summary resolution. Secondly, to quantify the impacts (positive or negative) of moving to open access, recorders were asked about their likely behaviour towards the schemes if all records were made fully accessible. Four options were available; withhold future records from the schemes, blur the resolution of future contributed records, continue to participate as before, and increase support for the schemes by submitting more records.

All four questions were obligatory and the survey was anonymous. The questionnaire was an online survey designed using DotMailer (www.dotmailer.com). In late May 2017, the online questionnaire was promoted to recorders by the UK regional co-ordinators. It remained live for just over two weeks with data being extracted on 13 June 2017.

*Analysis*

For each questionnaire, analysis was carried out on the aggregated responses but also separately after categorizing respondents by geographic or taxonomic interest, to test for differences between citizen scientists in different constituent countries of the UK (England, Scotland, Wales; Northern Ireland could not be tested separately due to a low sample size of responses to both questionnaires) and between recorders of butterflies, moths and both taxa. In addition, for the regional co-ordinator questionnaire data, we divided respondents into promoters, neutrals (passives) and detractors on the basis of their general support (on a 10-point scale) for open access to butterfly and moth records, using a slightly modified Net Promoter Score (NPS) methodology (Reichheld 2003; Keiningham *et al*. 2007). We classified those who scored 9 or 10 as promoters of open access, those who scored 5-8 as neutral and those scoring 1-4 as open access detractors. In standard NPS classification, scores as high as 6 are designated as detractors, but we increased the neutral segment to reflect better the range of views of our respondents. Categorising in this way enabled us to compare the opinions of regional co-ordinators with different levels of overall support for the principle of open access to the questions about specific details of record resolution, temporal delays in data release and species threat status.

Each comparison was analysed initially using Pearson Chi-squared and linear-by-linear association tests (Agresti 2002), accounting for the presence of ordinal variables. Where significant associations were found, cumulative link models with logit link were fitted, then Tukey-adjusted pairwise differences were investigated via least-squares means (LSM). All analyses were undertaken in R version 3.5.1 (R Core Team 2018) using the packages ordinal (Christensen 2018), coin (Hothorn *et al*. 2008) and emmeans (Lenth 2018). Goodness of fit for the cumulative link models was checked using likelihood ratio tests (nominal_test and scale_test in the ordinal package), in particular to assess whether the proportional odds assumption was satisfied. In some cases this assumption was not met, suggesting that the cumulative link model may not be appropriate, and in these instances pairwise differences among the explanatory variables were either assessed using the Cochran-Armitage test (with *p* values adjusted to account for false discovery rate) or only considered on the basis of summary statistics and figures.

### *Ethics statement*

Butterfly Conservation conforms strictly to appropriate legislation and codes of conduct relating to personal data and both questionnaires were designed and implemented in this context. For the regional co-ordinator questionnaire, full informed consent was obtained from all participants for the use of anonymised, aggregated responses in this research paper. Participants consented to the secure storage of data and access to the data by Butterfly Conservation employees involved in its analysis, and to publication of the arising results, for a period of five years, after which the data will be destroyed. Regional co-ordinator responses were anonymized prior to analysis. The online recorder questionnaire was completely anonymous and no personal data were collected. Participation in the questionnaires was voluntary and respondents were informed that the purpose was to gather views relating to open access to UK butterfly and moth occurrence data to aid the ongoing management and development of recording scheme data by Butterfly Conservation and other citizen science organisers.

## Results

### *Regional co-ordinators*

#### *Survey coverage*

Completed questionnaires were received from 104 regional co-ordinators representing response rates of 69% for the BNM and 68% for the NMRS networks. Responses were received from all four UK countries: 60 England, 2 Northern Ireland, 28 Scotland, 14 Wales.

#### *Support for open access*

Using our modified NPS scale, 39.8% of 103 regional co-ordinators who responded to this question were classified as open access promoters, 43.7% as neutrals and 16.5% as detractors. There was no difference in NPS value between respondents responsible for butterfly records, moth records and those who cover both taxa ($\chi^2 = 3.257$, df = 2, $p = 0.196$), although regional co-ordinators for butterflies generally appeared to have more moderate NPS values than other co-ordinators, with smaller proportions in both the promoter and detractor classes (Fig. A6.1).

Levels of general support for open access (measured with NPS) varied significantly between countries (Fig. 6.1, $\chi^2 = 9.766$, df = 2, $p = 0.008$); regional co-ordinators in Scotland were more in favour of open access than their counterparts in England (England – Scotland contrast: LSM estimate = -0.485, z ratio = -3.252, $p = 0.003$). Respondents from Wales had similar NPS scores to those from Scotland, but the difference with England was not statistically significant (England – Wales contrast: LSM estimate = -0.364, z ratio = -1.852, $p = 0.153$).

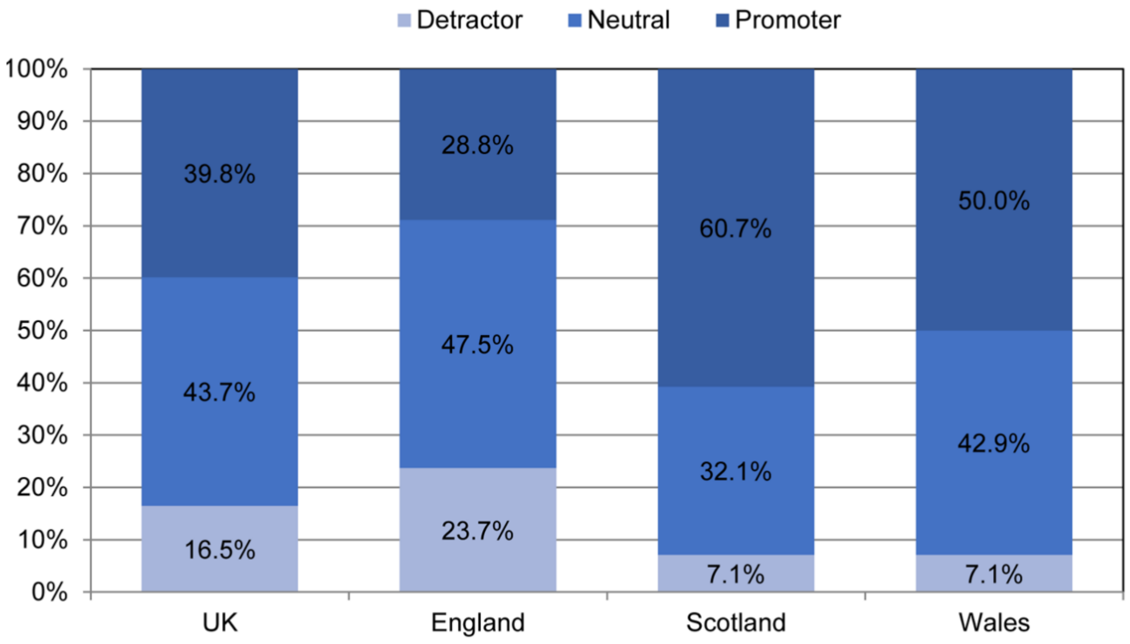

**Fig. 6.1** Levels of general support for open access, assessed by modified NPS categories, among regional co-ordinators from UK, England, Scotland and Wales (Northern Ireland not shown separately due to low sample size).

*Spatial resolution of records*

For records of threatened species, only 6.7% of the 104 regional co-ordinators were in favour of open access at full capture resolution (Fig. 6.2a). The majority (54.8%) preferred records of such species to be accessible only at 10km x 10km square (hereafter ‘10km square’) scale, the coarsest resolution offered in the questionnaire, with a further 29.8% in favour of 2km x 2km square (hereafter ‘2km square’) scale. Attitudes were very different for records of widespread species. For these, 37.5% of regional co-ordinators were in favour of open access at full capture resolution, while a further 40.4% supported open access at 1km x 1km square (hereafter ‘1km square’) resolution and 17.3% chose the 2km square scale (Fig. 6.2b). Only 4.8% (5 of 104 respondents) preferred the coarsest resolution option (10km square) for records of widespread species. These results provide support for our hypotheses, suggesting that fear of ecological damage underlies regional co-ordinators’ concerns about open access (as they were much more restrictive about records of threatened species than widespread ones) and also that these concerns can be ameliorated by blurring the spatial resolution of accessible records.

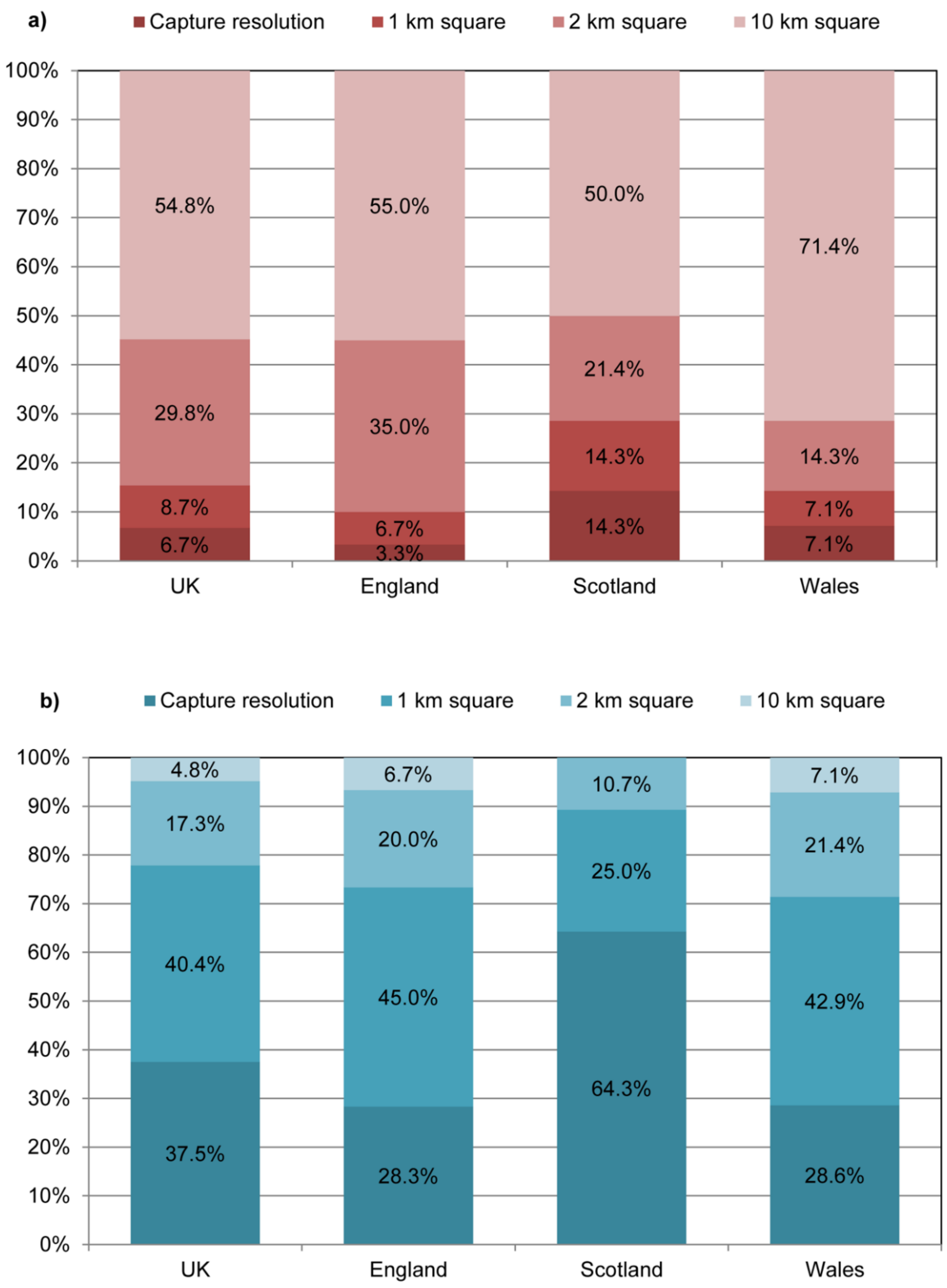

**Fig. 6.2** Preferred resolution of open access records of a) threatened species and b) widespread species among regional co-ordinators from UK, England, Scotland and Wales (Northern Ireland not shown separately due to low sample size).

For threatened species records there was no apparent difference between the responses from regional co-ordinators in different countries ($\chi^2$ = 3.364, df = 2, $p$ = 0.186), but there was a significant difference for widespread species ($\chi^2$ = 9.513, df = 2, $p$ = 0.009); regional co-ordinators in Scotland favoured finer resolution of open access records of widespread species than those in England (Scotland – England contrast: LSM estimate = -0.585, z ratio = -3.493, $p$ = 0.001) (Fig. 2b). There was also a tendency for regional co-ordinators in Scotland to favour finer resolution access than those in Wales (Scotland – Wales contrast: LSM estimate = -0.604, z ratio = -2.298, $p$ = 0.056). For example, in Scotland, 64.3% supported capture resolution access for widespread species compared to 28.3% in England and 28.6% in Wales.

There was a significant negative relationship between NPS category and preferred spatial resolution for both threatened (linear-by-linear association test Z = -3.794, $p$ = 0.0001) and widespread species (Z = -5.197, $p$ = <0.0001), with detractors favouring the coarsest resolutions. For records of widespread species, detractors favoured a coarser resolution than both neutrals (detractors – neutrals contrast: LSM estimate = 0.995, z ratio = 4.358, $p$ = <0.0001) and promoters (detractors – promoters contrast: LSM estimate = 1.396, z ratio = 6.207, $p$ = <0.0001), and neutrals favoured a coarser resolution than promoters (neutrals – promoters contrast: LSM estimate = 0.401, z ratio = 2.690, $p$ = 0.020).

Based on the goodness-of-fit tests, the cumulative link model was not reliable for pairwise contrasts between NPS categories and preferred spatial resolution for records of threatened species, but the responses suggest that detractors favoured coarser resolutions than neutrals who, in turn, favoured coarser resolutions than promoters (Fig. A6.2). None of the regional co-ordinators who classified as detractors or neutrals and only 17.1% of promoters were in favour of capture resolution open access for threatened species records. Even at the 2km square scale, only 17.6% detractors and 44.4% of neutrals were supportive, compared to the majority (58.5%) of promoters who were in favour of open access to records of threatened species at this resolution or even finer. In contrast, all of the regional co-ordinators classified as promoters or neutrals were in favour of open access to widespread species records at 2km square resolution, along with 70.6% of detractors. However, even for records of widespread species there was only limited support for full resolution open access, with 61.0% promoters, 28.9% neutrals and just 5.9% detractors (corresponding to one respondent) in favour.

The preferred spatial resolution of open access records of threatened species differed between regional co-ordinators covering butterflies, moths or both taxa ($\chi^2 = 9.376$, df = 2, $p = 0.009$) but there was no apparent difference for widespread species ($\chi^2 = 0.852$, df = 2, $p = 0.653$). Regional co-ordinators for butterflies preferred finer resolution open access for threatened species records than their moth counterparts (butterfly co-ordinators – moth co-ordinators contrast: LSM estimate = -0.627, z ratio = -3.441, $p = 0.002$) or for those covering both taxa (butterfly co-ordinators – co-ordinators of both taxa contrast: LSM estimate = -0.676, z ratio = -3.101, $p = 0.006$). Only 28.9% of regional co-ordinators for butterflies considered that the coarsest resolution (10km square) was required for open access records of threatened species, while 68.1% of regional co-ordinators for moths felt this was the appropriate resolution, as did 73.7% of co-ordinators responsible for both taxa.

### *Time lags*

Of the 100 regional co-ordinators that responded to the question about time lags, 74 favoured no delay to records being made open access, 21 supported a 5-year lag, 1 a 10-year lag and 4 a 20-year lag. NPS was significantly related to time lag (linear-by-linear association test Z = -5.351, $p = <0.0001$), with higher NPS correlated with shorter time lags. We were unable to undertake pairwise comparisons between NPS categories and time lags as the models did not satisfy goodness-of-fit tests. However, the significant relationship supports our hypothesis that concerns about open access can be lessened by deferring the release of records, at least among those regional co-ordinators who are generally more concerned about open access.

There was no apparent difference in the responses on time lags between regional co-ordinators covering different taxa ($\chi^2 = 2.371$, df = 2, $p = 0.306$), but there was between countries ($\chi^2 = 8.495$, df = 2, $p = 0.014$); only 11% (3 of 28 respondents) of regional co-ordinators in Scotland and 8% (1 of 13 respondents) in Wales advocated any time lag at all, and all of these were at the 5-year level, while 39% of 57 respondents in England were in favour of a delay in the release of records, including 9% who supported at least a 10-year delay. The difference in opinion on time lags was statistically significant (at the 5% level) between regional co-ordinators in England and Scotland (Cochran–Armitage test Z = 2.403, $p = 0.049$), but not between England and Wales (Cochran–Armitage test Z = 1.780, $p = 0.113$).

*Additional restrictions for species or colonies*

70.1% of 97 regional co-ordinators who answered this question stated that no additional restrictions on open access were required for species and/or sites in their area beyond those provided by constraints on spatial resolution and time lags.

*Creative Commons licences*

Of the 103 regional co-ordinators who answered this question, 79.6% favoured the Attribution-NonCommercial licence (CC-BY-NC), the most restrictive of the three Creative Commons licence options offered on the questionnaire. Only 3.9% of respondents selected the most open licence option (CC0).

Opinions about Creative Commons licences differed between countries ($\chi^2 = 8.105$, df = 2, $p = 0.017$). 46.2% of regional co-ordinators in Wales favoured the more open licences (CC0 and CC-BY), compared to 21.4% in Scotland and just 15.0% in England, but none of the pairwise comparisons were statistically significant (at the 5% level) using cumulative link models. There was no difference in views on Creative Commons licences between regional co-ordinators responsible for different taxa ($\chi^2 = 0.659$, df = 2, $p = 0.719$).

*Recorders*

*Survey coverage*

A total of 510 people completed the online questionnaire aimed at contributors of occurrence records to the BNM and NMRS. 25.3% identified as butterfly recorders, 25.5% as moth recorders and 49.2% stated that they recorded both groups. 367 (72.0%) respondents record mainly in England, 80 (15.7%) in Scotland, 58 (11.4%) in Wales and 5 (1.0%) in Northern Ireland.

*Spatial resolution of open access to own records*

32.7% of respondents preferred full open access, opting for public access to all their records at capture resolution. A further 50.8% indicated that they were happy for their records of widespread species (but not those of scarce/ threatened species) to be available at full capture resolution. Thus, for widespread species, 83.5% of respondents supported open access at capture resolution. In contrast, 16.5% of citizen scientists opposed capture resolution open access to any of their records (i.e. the spatial resolution of all records should be blurred to obscure precise locations), along with the 50.8% of respondents who thought that their records of scarce/threatened species should be blurred. Thus, 67.3% of respondents were against open access at capture resolution for some of their records. There were no significant differences between the views of recorders of different taxa ($\chi^2 = 2.022$, df = 2, $p = 0.364$) or in the different countries ($\chi^2 = 2.324$, df = 2, $p = 0.313$). The overall pattern, with a majority of recorders preferring to have their records of scarce/threatened species blurred but those of widespread species available at capture resolution provides further support for our two hypotheses; concern about ecological harm resulting from open access appears to be widespread among recorders and can be reduced by blurring the spatial resolution of records that are made publicly accessible.

*Future support for open access recording schemes*

The majority of respondents (76.7%) indicated that their participation in the projects would be affected positively (4.5% would provide more records) or unaffected (72.2%) if all records were made open access in full detail. In contrast, the results suggest that the participation in the recording schemes of 23.3% of respondents would be detrimentally impacted, either due to them reducing the precision of the records they submit (21.2%) or withholding records entirely (2.2%). There were no significant differences in responses between countries ($\chi^2 = 1.267$, df = 2, $p = 0.531$) or between recorders of different taxa ($\chi^2 = 2.393$, df = 2, $p = 0.302$).

## Discussion

We have shown that while there are high levels of support in principle for open access among UK citizen scientists that contribute, collate and verify

Lepidoptera occurrence data, they do not endorse full capture resolution open access nor unrestricted use of such data. Among the two groups of citizen scientists surveyed, only 6.7% of regional co-ordinators and 32.7% of recorders stated that records of all butterfly and moth species (widespread and threatened) should be open access at capture resolution, and 79.6% of regional co-ordinators felt that data reuse should be limited to non-commercial purposes. These findings are broadly similar to those in the only other study of citizen scientists' opinions that we are aware of; Ganzevoort *et al*. (2017) surveyed the demographics, motivations and views on data ownership and sharing of nearly 2,200 volunteer biodiversity recorders in the Netherlands. They found that only 12.3% of biodiversity recorders in the Netherlands supported unconditional reuse of their data, while 36.7% were opposed to commercial use of their data.

Current limitations to access and reuse of citizen science data are often attributed to the scientists or organisations running citizen science projects, who may face a range of technological, economic and cultural barriers and disincentives to data sharing (Reichman *et al*. 2011; Schmidt *et al*. 2016; Groom *et al*. 2017; Pearce-Higgins *et al*. 2018). However, our UK results and those from the Netherlands suggest that some limitation is in accordance with the wishes and expectations of citizen science participants.

### *Citizen scientist support for open access*

Despite data quality concerns (Kosmala *et al*. 2016; Aceves-Bueno *et al*. 2017; Specht & Lewandowski 2018), citizen science has great potential to address pressing matters in biodiversity monitoring, conservation and research (Theobald *et al*. 2015; Chandler *et al*. 2017; Pocock *et al*. 2018). Open access to citizen science data would maximise this potential through increased reuse and the application of new 'big data' techniques and cross-disciplinary studies (Culina *et al*. 2018b; Farley *et al*. 2018; Ma *et al*. 2018; Tulloch *et al*. 2018), as well as yielding benefits of increased transparency and public trust in science (Soranno *et al*. 2015).

Surveys of citizen scientists' motivations suggest support for these goals, with factors such as contributing to biodiversity conservation and science ranking highly (Hobbs & White 2012; Wright *et al*. 2015; West & Pateman 2016;

Ganzevoort *et al*. 2017; Lewandowski & Oberhauser 2017). In keeping with this, our surveys of attitudes among UK citizen scientists suggest general support for open access, albeit with some concern about threatened species. 39.8% of UK regional co-ordinators were classified as promoters of open access on the basis of NPS, with a further 43.7% as neutrals, and support was stronger in some UK countries (60.7% promoters in Scotland and 50.0% promoters in Wales). Among the much larger group of recorders, 32.7% felt that all their records should be open access at capture resolution and 76.7% indicated that they would maintain or enhance their participation if the data were to be made completely open. Considering just records of widespread species, 37.5% of regional co-ordinators and 83.5% of recorders were in favour of open access at capture resolution, with the proportion of regional co-ordinators in favour rising to 77.9% if records were restricted to 1km square resolution. In their survey of Dutch biodiversity recorders, Ganzevoort *et al*. (2017) also found evidence of general support for open access; 76.1% of citizen scientists regarded the data they contributed as a public good or as belonging to the organisation running the recording scheme i.e. they did not consider the data to be their personal property.

### *Concerns and alleviating factors*

Set against this general desire for data to be available and utilised are clear signals from our results and from other studies of concern regarding inappropriate use (Pearce-Higgins *et al*. 2018). As we did not ask participants about the motivations underlying their opinions on open access, discussion of their concerns must be speculative. It is well established that many citizen scientists want their records to contribute towards biodiversity conservation (e.g. Hobbs & White 2012; Lewandowski & Oberhauser 2017) but may be concerned that open access to data will undermine this goal. Threats to species (e.g. collectors targeting rare species, deliberate habitat destruction by landowners to avoid conservation responsibility/land-use restrictions, accidental damage to sites by naturalists wanting to see scarce species) are real (Tulloch *et al*. 2018), but the levels of perceived risk are subjective and individualistic. Such concerns may also engender support for licences that prohibit commercial reuse; citizen scientists appear to support uses of their data that are likely to benefit biodiversity conservation, but not those that are thought to cause harm (Ellis &

Waterton 2005; Ganzevoort *et al*. 2017). The perceived commodification of volunteer-gathered records, which runs counter to the traditional culture of data exchange within natural history, and a lack of transparency and feedback about the onward uses of the data may also contribute to restrictive attitudes towards licensing (Ellis & Waterton 2005). Other concerns may exist around privacy and the potential malicious use of personal information (e.g. names and locations of recorders) derived from species occurrence data (Bowser *et al*. 2014).

We extended the previous study by Ganzevoort *et al*. (2017) to gain a more nuanced understanding of these concerns and explored how citizen scientists' attitudes to open access were moderated by variation in spatial and temporal factors. We hypothesised that if concerns about open access related to potential damage to individual organisms, populations and habitats, then citizen scientists would be more restrictive with records of threatened species than widespread ones. Additionally, we posited that restricting the spatial resolution of publicly accessible data or delaying the release of data may both be expected to reduce the perceived risk. Other commonly raised fears around the personal privacy of the recorders themselves and of private land where charismatic species are present (which may be subject to trespass if the precise locations are made public) might also be ameliorated by such restrictions.

We found strong evidence to support both our hypotheses. There was a clear effect of spatial scale on attitudes to open access for UK Lepidoptera records (but not for the use of deferred release of data i.e. time lags). 37.5% of regional co-ordinators were in favour of capture resolution open access for records of widespread species and this rose cumulatively as the spatial scale was coarsened, such that 77.9% were in favour at 1km square resolution and 95.2% in favour at 2km square resolution. The impact of spatial resolution on open access opinions was even more pronounced when considering records of threatened species; regional co-ordinators were more cautious, with only 6.7% in favour at capture resolution, rising cumulatively to 15.4% at 1km square and 45.2% at 2km square resolution. Similar patterns were found when regional co-ordinators were grouped by general levels of support for open access (NPS categories) and each analysed separately.

The survey of recorders also suggested that spatial scale was an important factor in citizen scientists' attitudes towards open data. Generally, recorders were more supportive than regional co-ordinators of open access at capture resolution. Nevertheless, two-thirds (67.3%) of recorders felt that some (i.e. threatened species) or all of their records should be blurred to a coarser resolution than capture level for open access.

Therefore, although we did not attempt to determine the rationale underlying the opinions of citizen scientists, these results support both our hypotheses. The greater unwillingness to release records of threatened species at full capture resolution compared to records of widespread species suggests that the main concerns of citizen scientists relate to potential negative ecological impacts, rather than unease about privacy, confidentiality or intellectual property rights. Second, for the majority of contributors these concerns can be alleviated by blurring spatial location information. Interestingly, most respondents did not support deferral of the open release of records in addition to spatial restrictions, although 26.0% were in favour of a delay of at least five years.

### *Differences between roles, countries and taxa*

The differing nature of the roles of regional co-ordinators and recorders and the fact that they were asked different questions makes it inappropriate to undertake a direct statistical comparison of their views. In addition, it is probable that some regional co-ordinators also completed the recorder questionnaire and so the two samples may not be independent. The findings on spatial resolution suggest, however, that the regional co-ordinators were more restrictive, on average, than recorders in their attitudes to open access. Further work is required to elucidate the causes of the seemingly greater risk aversion among regional co-ordinators, as our questionnaires did not examine the reasons underlying stated opinions. They may stem from complex combinations of ecological (e.g. increased awareness of possible threats to species), legal (e.g. concerns about acts of trespass and personal data under the General Data Protection Regulation), personal (e.g. greater time investment in the data), ethical (e.g. a sense of responsibility as custodians of records contributed by other citizen scientists) and cultural (e.g. traditional principles of data exchange in natural history) considerations. The latter may be particularly important given that regional co-ordinators are amateur expert naturalists, whereas recorders are a much more diverse group ranging from committed amateur naturalists to complete beginners (e.g. see Everett & Geoghegan 2016). Irrespective of the causes, however, if restrictions on open access to recording scheme data, informed by the views of regional co-ordinators, are contrary to the wishes of most citizen scientist participants, this may risk demotivation, loss of support and, ultimately, reduced levels of species recording.

Significant differences were found between the opinions of regional co-ordinators in England and Scotland. Regional co-ordinators in Scotland had higher NPS values than their counterparts in England, indicating greater support in general for the principles of open access to Lepidoptera occurrence records. This predisposition was reflected in attitudes to more specific options, whereby regional co-ordinators in Scotland favoured finer spatial scale resolution of open access records for widespread species and shorter time lags before records are released than their colleagues in England.

The causes of these differences are not known and require further research. However, we speculate that two factors may contribute to these contrasting attitudes. First, long-term abundance trends of butterflies and moths differ geographically within the UK. The abundance of 337 species of widespread moths has decreased significantly in southern Britain (most of England and all of Wales) but not in northern Britain (Scotland plus part of northern England) (Conrad *et al*. 2006). Similarly, the abundance of wider countryside butterflies has decreased significantly in England but not in Scotland (Fox *et al*. 2015). Thus, regional co-ordinators in England, where greater declines have occurred, might be more sensitive to potential adverse effects on butterflies and moths arising from open access to data and this results in more restrictive attitudes than regional co-ordinators in Scotland.

Second, there are substantial differences between Scotland and the rest of the UK in the legal framework relating to public access to land. The Land Reform (Scotland) Act 2003 confers a public ‘right to roam’ over almost all land in Scotland, while similar rights (under the Countryside and Rights of Way Act 2000) cover only c.8% - 12% of the total land area of England and Wales

(Lovett 2012). The situation is even more restrictive in Northern Ireland. Regional co-ordinators in Scotland may have reduced concerns, therefore, compared to their counterparts in other UK countries, about either exposing acts of trespass by recorders or inadvertently encouraging others to trespass on private land (thereby undermining relationships between recorders and landowners) as a result of records being made open access.

Interestingly, the online survey of recorders found no significant differences between UK countries. This suggests that the differing attitudes of regional co-ordinators in England and Scotland relates to their roles as custodians of local datasets.

In contrast to the clear country-level differences, attitudes of regional co-ordinators varied very little depending on the taxon (butterflies, moths or both) for which they have responsibility. The only significant result in our analysis was that regional co-ordinators for butterflies favoured finer spatial resolution open access for records of threatened species than regional co-ordinators who cover moths or both taxa. Possible reasons for this might include that there are more UK populations of the most threatened butterflies than the most threatened moths, that sites for threatened butterflies are often well known already or that extra visitors to sites of threatened butterflies are likely to be less intrusive for landowners than those wanting to see threatened moths if the latter are nocturnal. There were no significant differences between the opinions of recorders based on taxon of interest.

### *Wider applicability*

The wider applicability of our findings depends on the representativeness of our sampling. With 69% and 68% response rates among regional co-ordinators, we can have high confidence that our results are representative of this key group of UK Lepidoptera-recording volunteers. However, we do not know how many people participate annually in the BNM and NMRS recording schemes, so we cannot measure the response rate for our online questionnaire aimed at recorders. While 510 responses is reasonable for statistical analysis, it likely represents only a small proportion of the total number of citizen science contributors to these projects. In addition, the sample is likely to be biased, as the online survey was not distributed randomly or systematically but promoted to recorders by the regional co-ordinators. This clearly limits our ability to generalise from the findings.

Another limitation stems from variation between participants. Analyses of this variation have classified citizen scientists by expertise in species identification and by temporal and spatial patterns of participation in particular projects (Ponciano & Brasileiro 2014; Boakes *et al*. 2016; Everett & Geoghegan 2016; Johnston *et al*. 2018). Boakes *et al*. (2016), for example, categorised citizen scientists undertaking biodiversity recording as ‘dabbler’, ‘steady’ or ‘enthusiast’ depending on their temporal participation, while Everett & Geoghegan (2016) utilised a continuum of engagement, on the basis of past involvement in natural history. While all citizen scientists can contribute useful data, their motivations and strength of commitment to particular projects vary considerably between individuals and also over time for individuals. It is likely that attitudes towards open access to citizen science data would also vary between individuals and over time, and might covary with other metrics describing the engagement behaviour of citizen scientists. By definition, given their role and responsibilities to the BNM and NMRS projects, the regional co-ordinators that took part in our study are highly motivated, committed and knowledgeable volunteers, many of whom have a passion for biodiversity conservation. Their views on open access are of fundamental importance for the ongoing development of the BNM and NMRS projects, but cannot reasonably be generalised to the thousands of citizens who participate to a greater or lesser extent in the schemes. Similarly, as the recorders who responded to our online questionnaire were not selected at random, it is likely that these may also be a biased sample, with views on open access that might differ from less active or more recent participants.

Even within our sampled audience of citizen scientists, we found evidence of differences in attitude towards open access between countries. Whatever the causes, this variation within the UK suggests that there will also be differences between the UK and other countries. This limits the applicability of our results but stresses the importance of seeking the opinions of and establishing dialogue with citizen scientists on this issue, rather than making assumptions.

### *Practical recommendations for citizen science*

A key factor in the creation of a citizen science project is the development of a comprehensive yet clear data policy (James 2011). This needs to take into account not only the requirements of the project itself, and its aspirations for future data sharing and scientific publication, but also any legal requirements for open access imposed by funding organisations. For example, in the UK Butterfly Monitoring Scheme, a systematic monitoring programme run by Butterfly Conservation and partner organisations, it is a condition of long-standing financial support from government departments and agencies that data are made freely available under an Open Government Licence. Schemes such as those addressed in this study, which are not bound by funder requirements regarding open data, provide an opportunity to plan data access in the light of contributors' attitudes. A data policy must, of course, also comply with relevant legislation relating to the protection of personal data, such as the European Union's General Data Protection Regulation. The use of widely recognised licences, such as Creative Commons licences, is recommended to ensure clarity for both participants and prospective data users, as well as compatibility with other projects and data repositories (e.g. the Atlas of Living Australia, www.ala.org.au) locally and globally. Most importantly, we recommend that any data policy developed for a citizen science project should be actively disseminated to potential contributors to ensure that they are aware of the uses to which their data will be put and are therefore able to make an informed choice prior to participation.

Despite its limitations, our study provides useful information on the development of open access data policies that is of wider relevance to biodiversity citizen science projects. In particular, the heterogeneity of views present in these relatively small samples shows that organisers would be well-advised to consult with potential participants on matters of data access in advance as part of project development. Similarly, funding organisations, statutory agencies and policy makers may wish to reflect on the diversity of views revealed by our questionnaires, and previous studies (e.g. Ellis *et al*. 2005), in their drive for open citizen science data. Our results suggest that the cultural context is likely to be extremely important in influencing attitudes to open access among citizen scientists; not only are these likely to differ substantially between nationalities, but also between participants with different roles in projects and levels of past engagement with natural history and citizen science.

## Conclusions

In order to maximise the scientific and societal benefits of citizen science, the views and motivations of participants must be considered. Our study shows that, contrary to common assumptions, UK citizen scientists taking part in butterfly and moth recording have diverse, in some cases polarised, views on open access and there was substantial variation between different countries and between volunteers with different roles. Overall, many participants are supportive, in principle, of open access to the data they gather, but are mindful of possible negative ecological impacts that may result. Our results suggest that the majority of participants favour increasing access to these data, and that the concerns of many could be ameliorated by limiting the spatial resolution of open records, particularly of threatened species, and licencing reuse for non-commercial purposes. Globally, citizen science schemes have great potential to help address the enormous challenges facing biodiversity, but to do so effectively, must be responsive to the changing attitudes and new opportunities afforded by open data.

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## Appendix 1: Richard Fox Peer-reviewed publications

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## Appendix 3: Supporting Information for Chapter 3: Long-term changes to the frequency of occurrence of British moths are consistent with opposing and synergistic effects of climate and land-use changes

### Frescalo analysis of distribution change

Frescalo is a method that can be applied to species occurrence data to assess variation in recorder effort and produce trends in species frequency over time (Hill 2012). The method requires two parameters to be identified by users, although sensitivity analyses suggest that precise values are not critical (see supporting Information in Hill 2012). The first parameter is the standard neighbourhood frequency Φ, which generally reflects how well the species group is recorded. The default value of Φ is 0.74, but the Frescalo program provides an output value of phi using a convergence algorithm which scales local neighbourhood frequencies by sampling effort multipliers. For groups that are not completely recorded, Hill (2012) suggests setting Φ greater than the 98th percentile of observed values of local neighbourhood frequency. Therefore, for the analysis of the entire Great Britain (GB) macro-moth dataset we increased the value of Φ to 0.89, to remain above this 98th percentile. For the analysis where northern and southern halves of geographically widespread species distributions were assessed separately we used a Φ value of 0.94 for southern hectads (10km x 10km grid squares), which were relatively better recorded than northern hectads, where we set the value of Φ to 0.74.

The second input parameter required by Frescalo is the R*, the proportion of species treated as benchmark species. The most common species in a local neighbourhood are used as ‘benchmarks’ to give an indication of how well recorded a given hectad is, and this is then used to modify the reporting frequency of the focal species in order to inform on the probability of occurrence. Ideally, the benchmark species should be relatively stable in frequency over time, but sensitivity analyses have shown that the method is robust to dynamic benchmark species (see Supporting Information in Hill 2012). We retained the default value of 0.27 for this parameter.

Following the methods in Hill (2012), the probability of reporting species in a given hectad in a given time period depends on 1) recorder effort, which is measured as the proportion of benchmark species reported from the hectad in that time period, 2) the time-independent probability of the species occurring in that hectad and 3) the Relative Reporting Rate (RRR) for the time period. The RRR is the ratio of the rate at which the focal species is reported to the rate at which the benchmark species are reported within the range of occupancy of the focal species in a given time period. In order that values of RRR for rare species are of comparable magnitude to those for commoner species, this is relativized by dividing by the time-independent probability of finding the focal species, also within its range of occupancy. Relative Reporting Rate is obtained iteratively through the formula given in Hill (2012). Hence, although it is a measure without units, the change in Relative Reporting Rate (ΔRRR) between time periods describes temporal changes in the estimated species frequency across all hectads, i.e. a negative ΔRRR indicates a decline in species frequency.

As described in Chapter 3, the significance of these trends can be determined by:

$$z = \frac{t_2 - t_1}{\sqrt{{\sigma_2}^2 + {\sigma_1}^2}}$$

where $t_1$ and $t_2$ are the Relative Reporting Rates of a given species from the first and second time periods and $\sigma_1{}^2$ and $\sigma_2{}^2$ are the variances associated with the RRR for periods $t_1$ and $t_2$ respectively.

The Frescalo method estimates species' frequency of occurrence, which is a function of both local abundance and distribution extent. In practice, these two variables are inextricably linked because the probability of a species being recorded in a grid square depends both on local abundance and on recording effort. In theory, changes in frequency of occurrence could relate solely to variation in local abundance with no distribution change. However, because we aggregated data over broad time periods in our study (thereby increasing recording effort), and the fact that changes in frequency of occurrence calculated for macro-moths correlated significantly with the raw number of occupied grid squares (Figs A3.3 & A3.4), it is likely that changes in frequency described here also reflect changes in moth distribution extents in GB.

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**Table A3.1** Change in frequency of occurrence (measured as annual change in relative reporting rate; ∆ RRR year$^{-1}$) 1970-99 versus 2000-10 for 673 resident GB macro-moths.

| **Taxon** | **∆ RRR year$^{-1}$** | **z** | **Signif** |
|---|---|---|---|
| *Abraxas grossulariata* | -0.0085 | -5.40 | p<0.05 |
| *A. sylvata* | -0.0040 | -3.98 | p<0.05 |
| *Abrostola tripartita* | 0.0139 | 6.81 | p<0.05 |
| *A. triplasia* | -0.0002 | -0.22 | NS |
| *Acasis viretata* | 0.0083 | 7.66 | p<0.05 |
| *Achlya flavicornis* | 0.0007 | 0.82 | NS |
| *Acronicta aceris* | 0.0078 | 4.94 | p<0.05 |
| *A. alni* | -0.0015 | -1.62 | NS |
| *A. euphorbiae* | -0.0013 | -0.54 | NS |
| *A. leporina* | 0.0022 | 2.19 | p<0.05 |
| *A. megacephala* | 0.0036 | 2.82 | p<0.05 |
| *A. menyanthidis* | -0.0055 | -4.52 | p<0.05 |
| *A. psi* | -0.0071 | -6.58 | p<0.05 |
| *A. rumicis* | 0.0016 | 1.16 | NS |
| *A. tridens* | -0.0051 | -5.25 | p<0.05 |
| *Actebia praecox* | -0.0076 | -4.78 | p<0.05 |
| *Adscita geryon* | -0.0052 | -2.24 | p<0.05 |
| *A. statices* | -0.0041 | -3.60 | p<0.05 |
| *Aethalura punctulata* | -0.0044 | -4.74 | p<0.05 |
| *Agriopis aurantiaria* | -0.0054 | -6.77 | p<0.05 |
| *A. leucophaearia* | -0.0011 | -1.30 | NS |
| *A. marginaria* | -0.0038 | -3.78 | p<0.05 |
| *Agrochola circellaris* | -0.0010 | -1.05 | NS |
| *A. helvola* | -0.0064 | -7.87 | p<0.05 |
| *A. litura* | -0.0061 | -5.67 | p<0.05 |
| *A. lota* | 0.0024 | 2.39 | p<0.05 |
| *A. lychnidis* | -0.0038 | -3.01 | p<0.05 |
| *A. macilenta* | 0.0033 | 3.43 | p<0.05 |
| *Agrotis cinerea* | -0.0082 | -5.50 | p<0.05 |
| *A. clavis* | 0.0016 | 1.46 | NS |
| *A. exclamationis* | -0.0109 | -3.73 | p<0.05 |
| *A. puta* | 0.0100 | 5.05 | p<0.05 |
| *A. ripae* | -0.0017 | -0.72 | NS |
| *A. segetum* | -0.0029 | -2.38 | p<0.05 |
| *A. trux* | -0.0045 | -1.72 | NS |
| *A. vestigialis* | -0.0103 | -6.37 | p<0.05 |
| *Alcis jubata* | -0.0016 | -1.14 | NS |
| *A. repandata* | -0.0018 | -0.66 | NS |
| *Aleucis distinctata* | -0.0116 | -2.59 | p<0.05 |
| *Allophyes oxyacanthae* | -0.0020 | -1.81 | NS |
| *Alsophila aescularia* | -0.0014 | -1.20 | NS |
| *Amphipoea crinanensis* | -0.0036 | -2.36 | p<0.05 |
| *A. fucosa* | -0.0019 | -1.28 | NS |
| *A.lucens* | 0.0003 | 0.25 | NS |
| *A. oculea* | -0.0028 | -3.41 | p<0.05 |
| *Amphipyra tragopoginis* | -0.0160 | -11.14 | p<0.05 |
| *Anaplectoides prasina* | 0.0017 | 1.77 | NS |
| *Anarta cordigera* | -0.0249 | -3.33 | p<0.05 |
| *A. melanopa* | -0.0003 | -0.09 | NS |
| *A. myrtilli* | -0.0052 | -5.30 | p<0.05 |
| *Angerona prunaria* | -0.0042 | -2.87 | p<0.05 |
| *Anticlea badiata* | -0.0020 | -1.98 | p<0.05 |
| *A. derivata* | 0.0029 | 2.88 | p<0.05 |
| *Anticollix sparsata* | -0.0116 | -3.18 | p<0.05 |
| *Antitype chi* | -0.0080 | -7.53 | p<0.05 |
| *Apamea anceps* | -0.0020 | -1.57 | NS |
| *A. crenata* | 0.0000 | -0.03 | NS |
| *A. epomidion* | -0.0008 | -0.84 | NS |
| *A. furva* | -0.0074 | -6.37 | p<0.05 |
| *A. lithoxylaea* | -0.0042 | -2.88 | p<0.05 |

| | | | |
|---|---|---|---|
| *A. monoglypha* | -0.0105 | -1.63 | NS |
| *A. oblonga* | -0.0111 | -5.85 | p<0.05 |
| *A. ophiogramma* | -0.0003 | -0.37 | NS |
| *A. remissa* | -0.0032 | -2.51 | p<0.05 |
| *A. scolopacina* | 0.0031 | 2.96 | p<0.05 |
| *A. sordens* | -0.0064 | -5.59 | p<0.05 |
| *A. sublustris* | -0.0008 | -0.56 | NS |
| *A. unanimis* | 0.0009 | 1.03 | NS |
| *A. zeta* | -0.0013 | -0.54 | NS |
| *Apeira syringaria* | -0.0022 | -2.27 | p<0.05 |
| *Aplocera efformata* | -0.0025 | -2.04 | p<0.05 |
| *A. plagiata* | -0.0060 | -6.67 | p<0.05 |
| *Apocheima hispidaria* | -0.0021 | -1.69 | NS |
| *Apoda limacodes* | 0.0098 | 4.19 | p<0.05 |
| *Aporophyla australis* | -0.0014 | -0.39 | NS |
| *A. nigra* | 0.0080 | 7.34 | p<0.05 |
| *Archanara algae* | -0.0034 | -0.75 | NS |
| *A. dissoluta* | -0.0004 | -0.28 | NS |
| *A. geminipuncta* | 0.0031 | 2.15 | p<0.05 |
| *A. sparganii* | 0.0110 | 5.40 | p<0.05 |
| *Archiearis notha* | -0.0045 | -2.37 | p<0.05 |
| *A. parthenias* | -0.0006 | -0.57 | NS |
| *Arctia caja* | -0.0161 | -10.23 | p<0.05 |
| *A. villica* | -0.0041 | -2.68 | p<0.05 |
| *Arenostola phragmitidis* | -0.0003 | -0.17 | NS |
| *Aspitates gilvaria* | -0.0039 | -0.88 | NS |
| *Asteroscopus sphinx* | -0.0020 | -1.98 | p<0.05 |
| *Asthena albulata* | -0.0009 | -0.88 | NS |
| *Atethmia centrago* | 0.0060 | 5.44 | p<0.05 |
| *Atolmis rubricollis* | 0.0134 | 12.30 | p<0.05 |
| *Autographa bractea* | -0.0063 | -4.92 | p<0.05 |
| *A. jota* | -0.0087 | -6.54 | p<0.05 |
| *A. pulchrina* | -0.0090 | -5.17 | p<0.05 |
| *Axylia putris* | 0.0060 | 2.69 | p<0.05 |
| *Bena bicolorana* | 0.0035 | 3.39 | p<0.05 |
| *Biston betularia* | 0.0028 | 1.22 | NS |
| *B. strataria* | 0.0018 | 1.69 | NS |
| *Blepharita adusta* | -0.0068 | -7.77 | p<0.05 |
| *Brachionycha nubeculosa* | 0.0053 | 0.95 | NS |
| *Brachylomia viminalis* | -0.0084 | -9.54 | p<0.05 |

| | | | |
|---|---|---|---|
| *Bupalus piniaria* | -0.0057 | -5.42 | p<0.05 |
| *Cabera exanthemata* | 0.0028 | 1.68 | NS |
| *Cabera pusaria* | -0.0042 | -1.89 | NS |
| *Callimorpha dominula* | 0.0033 | 1.82 | NS |
| *Callistege mi* | -0.0024 | -2.66 | p<0.05 |
| *Calliteara pudibunda* | 0.0142 | 6.92 | p<0.05 |
| *Calophasia lunula* | 0.0165 | 3.86 | p<0.05 |
| *Campaea margaritata* | 0.0076 | 2.81 | p<0.05 |
| *Camptogramma bilineata* | -0.0193 | -7.86 | p<0.05 |
| *Caradrina morpheus* | -0.0046 | -3.15 | p<0.05 |
| *Carsia sororiata* | -0.0035 | -2.21 | p<0.05 |
| *Catarhoe cuculata* | -0.0049 | -3.08 | p<0.05 |
| *C. rubidata* | -0.0029 | -2.18 | p<0.05 |
| *Catocala fraxini* | -0.0005 | -0.27 | NS |
| *C. nupta* | 0.0001 | 0.12 | NS |
| *C. promissa* | -0.0028 | -0.71 | NS |
| *Celaena haworthii* | -0.0046 | -3.76 | p<0.05 |
| *C. leucostigma* | -0.0018 | -1.95 | NS |
| *Cepphis advenaria* | -0.0037 | -1.41 | NS |
| *Cerapteryx graminis* | -0.0157 | -10.23 | p<0.05 |
| *Cerastis leucographa* | -0.0026 | -1.59 | NS |
| *C. rubricosa* | -0.0015 | -1.46 | NS |
| *Cerura vinula* | -0.0073 | -8.32 | p<0.05 |
| *Charanyca trigrammica* | 0.0068 | 5.30 | p<0.05 |
| *Charissa obscurata* | -0.0088 | -7.08 | p<0.05 |
| *Chesias legatella* | -0.0048 | -5.15 | p<0.05 |
| *C. rufata* | -0.0071 | -5.32 | p<0.05 |
| *Chiasmia clathrata* | -0.0093 | -7.53 | p<0.05 |
| *Chilodes maritimus* | 0.0009 | 0.62 | NS |
| *Chlorissa viridata* | -0.0075 | -3.19 | p<0.05 |
| *Chloroclysta citrata* | -0.0095 | -8.43 | p<0.05 |
| *C. concinnata* | -0.0156 | -2.64 | p<0.05 |
| *C. miata* | -0.0044 | -4.89 | p<0.05 |
| *C. siterata* | 0.0189 | 16.94 | p<0.05 |
| *C. truncata* | 0.0108 | 3.60 | p<0.05 |
| *Chloroclystis v-ata* | 0.0033 | 2.53 | p<0.05 |
| *Chortodes brevilinea* | -0.0100 | -1.12 | NS |

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|---|---|---|---|
| *C. elymi* | -0.0100 | -2.37 | p<0.05 |
| *C. extrema* | 0.0027 | 0.49 | NS |
| *C. fluxa* | -0.0047 | -1.84 | NS |
| *C. pygmina* | 0.0016 | 1.56 | NS |
| *Cidaria fulvata* | -0.0044 | -3.47 | p<0.05 |
| *Cilix glaucata* | -0.0020 | -1.21 | NS |
| *Cleora cinctaria* | -0.0039 | -1.10 | NS |
| *Cleorodes lichenaria* | 0.0033 | 2.60 | p<0.05 |
| *Clostera curtula* | 0.0058 | 4.40 | p<0.05 |
| *C. pigra* | -0.0075 | -6.11 | p<0.05 |
| *Coenobia rufa* | 0.0068 | 6.10 | p<0.05 |
| *Coenocalpe lapidata* | -0.0014 | -0.40 | NS |
| *Colocasia coryli* | 0.0068 | 5.43 | p<0.05 |
| *Colostygia multistrigaria* | -0.0013 | -1.47 | NS |
| *C. olivata* | -0.0069 | -5.59 | p<0.05 |
| *C. pectinataria* | 0.0130 | 4.04 | p<0.05 |
| *Colotois pennaria* | 0.0000 | -0.05 | NS |
| *Comibaena bajularia* | -0.0032 | -2.48 | p<0.05 |
| *Conistra ligula* | 0.0026 | 2.77 | p<0.05 |
| *C. rubiginea* | 0.0109 | 5.91 | p<0.05 |
| *C. vaccinii* | 0.0064 | 5.15 | p<0.05 |
| *Coscinia cribraria* | -0.0100 | -2.16 | p<0.05 |
| *Cosmia affinis* | -0.0043 | -4.01 | p<0.05 |
| *C. diffinis* | -0.0083 | -7.08 | p<0.05 |
| *C. pyralina* | -0.0051 | -3.94 | p<0.05 |
| *C. trapezina* | -0.0002 | -0.14 | NS |
| *Cosmorhoe ocellata* | -0.0060 | -3.91 | p<0.05 |
| *Cossus cossus* | -0.0026 | -2.81 | p<0.05 |
| *Craniophora ligustri* | 0.0112 | 9.60 | p<0.05 |
| *Crocallis elinguaria* | -0.0042 | -2.02 | p<0.05 |
| *Cryphia domestica* | 0.0004 | 0.30 | NS |
| *C. muralis* | 0.0037 | 1.96 | p<0.05 |
| *Cucullia absinthii* | -0.0065 | -4.06 | p<0.05 |
| *C. asteris* | -0.0087 | -3.91 | p<0.05 |
| *C. chamomillae* | -0.0056 | -5.87 | p<0.05 |
| *C. umbratica* | -0.0054 | -5.97 | p<0.05 |
| *Cybosia mesomella* | -0.0014 | -1.16 | NS |
| *Cyclophora albipunctata* | -0.0014 | -1.13 | NS |
| *C. annularia* | 0.0050 | 3.05 | p<0.05 |
| *C. linearia* | 0.0065 | 5.88 | p<0.05 |

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|---|---|---|---|
| *C. pendularia* | -0.0002 | -0.10 | NS |
| *C. porata* | -0.0076 | -6.34 | p<0.05 |
| *C. punctaria* | 0.0081 | 6.84 | p<0.05 |
| *Cymatophorima diluta* | -0.0038 | -3.14 | p<0.05 |
| *Dasypolia templi* | -0.0043 | -3.44 | p<0.05 |
| *Deilephila elpenor* | 0.0114 | 5.42 | p<0.05 |
| *D. porcellus* | 0.0056 | 5.58 | p<0.05 |
| *Deileptenia ribeata* | 0.0039 | 3.76 | p<0.05 |
| *Deltote bankiana* | 0.0054 | 0.98 | NS |
| *D. uncula* | -0.0063 | -4.33 | p<0.05 |
| *Diachrysia chrysitis* | -0.0043 | -1.68 | NS |
| *D. chryson* | -0.0027 | -0.87 | NS |
| *D. sannio* | -0.0018 | -1.41 | NS |
| *Diaphora mendica* | 0.0006 | 0.50 | NS |
| *Diarsia brunnea* | -0.0043 | -4.04 | p<0.05 |
| *D. dahlii* | -0.0059 | -5.81 | p<0.05 |
| *D. mendica* | -0.0089 | -3.78 | p<0.05 |
| *D. rubi* | 0.0074 | 3.39 | p<0.05 |
| *Dicallomera fascelina* | -0.0013 | -0.85 | NS |
| *Dichonia aprilina* | 0.0035 | 4.01 | p<0.05 |
| *Dicycla oo* | -0.0040 | -1.49 | NS |
| *Diloba caeruleocephala* | -0.0137 | -13.37 | p<0.05 |
| *Discestra trifolii* | -0.0057 | -5.13 | p<0.05 |
| *Discoloxia blomeri* | -0.0001 | -0.09 | NS |
| *Drepana falcataria* | 0.0029 | 2.35 | p<0.05 |
| *Drymonia dodonaea* | -0.0004 | -0.39 | NS |
| *D. ruficornis* | 0.0041 | 4.00 | p<0.05 |
| *Dryobotodes eremita* | 0.0013 | 1.41 | NS |
| *Dypterygia scabriuscula* | -0.0012 | -0.83 | NS |
| *Dyscia fagaria* | -0.0062 | -5.37 | p<0.05 |
| *Earias clorana* | 0.0080 | 4.85 | p<0.05 |
| *Ecliptopera silaceata* | 0.0074 | 4.11 | p<0.05 |
| *Ectropis bistortata* | 0.0049 | 3.62 | p<0.05 |
| *E. crepuscularia* | -0.0102 | -11.74 | p<0.05 |
| *Egira conspicillaris* | -0.0022 | -0.58 | NS |
| *Eilema caniola* | 0.0095 | 3.57 | p<0.05 |
| *E. complana* | 0.0082 | 5.38 | p<0.05 |
| *E. depressa* | 0.0190 | 15.51 | p<0.05 |

| | | | |
|---|---|---|---|
| *E. griseola* | 0.0235 | 13.11 | p<0.05 |
| *E. lurideola* | 0.0171 | 6.34 | p<0.05 |
| *E. pygmaeola* | 0.0061 | 0.94 | NS |
| *E. sororcula* | 0.0209 | 16.11 | p<0.05 |
| *Elaphria venustula* | 0.0024 | 0.92 | NS |
| *Electrophaes corylata* | -0.0034 | -3.18 | p<0.05 |
| *Ematurga atomaria* | -0.0070 | -6.44 | p<0.05 |
| *Enargia paleacea* | -0.0010 | -0.63 | NS |
| *Endromis versicolora* | -0.0065 | -1.63 | NS |
| *Ennomos alniaria* | 0.0014 | 0.94 | NS |
| *E. autumnaria* | -0.0016 | -0.76 | NS |
| *E. erosaria* | -0.0093 | -9.68 | p<0.05 |
| *E. fuscantaria* | 0.0035 | 2.78 | p<0.05 |
| *E. quercinaria* | -0.0033 | -3.54 | p<0.05 |
| *Entephria caesiata* | -0.0167 | -10.62 | p<0.05 |
| *E. flavicinctata* | -0.0011 | -0.33 | NS |
| *Epione repandaria* | -0.0008 | -0.84 | NS |
| *Epirrhoe alternata* | 0.0012 | 0.23 | NS |
| *E. galiata* | -0.0046 | -4.23 | p<0.05 |
| *E. rivata* | -0.0048 | -4.71 | p<0.05 |
| *E. tristata* | -0.0080 | -6.25 | p<0.05 |
| *Epirrita autumnata* | -0.0037 | -3.90 | p<0.05 |
| *E. christyi* | -0.0043 | -4.32 | p<0.05 |
| *E. dilutata* | -0.0055 | -6.29 | p<0.05 |
| *E. filigrammaria* | -0.0078 | -6.44 | p<0.05 |
| *Erannis defoliaria* | -0.0061 | -6.25 | p<0.05 |
| *Eremobia ochroleuca* | -0.0010 | -0.52 | NS |
| *Eriogaster lanestris* | -0.0015 | -1.32 | NS |
| *Euchoeca nebulata* | 0.0023 | 2.51 | p<0.05 |
| *Euclidia glyphica* | -0.0014 | -1.24 | NS |
| *Eugnorisma depuncta* | -0.0042 | -2.62 | p<0.05 |
| *E. glareosa* | -0.0006 | -0.53 | NS |
| *Eulithis mellinata* | -0.0094 | -10.23 | p<0.05 |
| *E. populata* | -0.0125 | -8.05 | p<0.05 |
| *E. prunata* | 0.0014 | 1.36 | NS |
| *E. pyraliata* | -0.0052 | -2.70 | p<0.05 |
| *E. testata* | -0.0070 | -6.92 | p<0.05 |
| *Euphyia biangulata* | 0.0001 | 0.06 | NS |

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|---|---|---|---|
| *E. unangulata* | -0.0040 | -3.20 | p<0.05 |
| *Eupithecia abbreviata* | 0.0079 | 7.25 | p<0.05 |
| *E. abietaria* | -0.0028 | -1.36 | NS |
| *E. absinthiata* | 0.0027 | 2.68 | p<0.05 |
| *E. assimilata* | 0.0039 | 4.70 | p<0.05 |
| *E. centaureata* | 0.0008 | 0.57 | NS |
| *E. distinctaria* | -0.0080 | -4.28 | p<0.05 |
| *E. dodoneata* | 0.0060 | 5.92 | p<0.05 |
| *E. egenaria* | 0.0060 | 1.35 | NS |
| *E. exiguata* | 0.0069 | 6.40 | p<0.05 |
| *E.expallidata* | -0.0064 | -4.14 | p<0.05 |
| *E. haworthiata* | -0.0001 | -0.07 | NS |
| *E. icterata* | -0.0101 | -9.26 | p<0.05 |
| *E. indigata* | 0.0016 | 1.61 | NS |
| *E. innotata* | -0.0064 | -5.23 | p<0.05 |
| *E. insigniata* | -0.0080 | -3.63 | p<0.05 |
| *E. inturbata* | 0.0003 | 0.29 | NS |
| *E. irriguata* | -0.0071 | -3.23 | p<0.05 |
| *E. lariciata* | -0.0033 | -3.63 | p<0.05 |
| *E. linariata* | -0.0032 | -3.15 | p<0.05 |
| *E. millefoliata* | 0.0058 | 2.18 | p<0.05 |
| *E. nanata* | -0.0050 | -4.95 | p<0.05 |
| *E. phoeniceata* | 0.0081 | 3.78 | p<0.05 |
| *E. pimpinellata* | -0.0079 | -4.72 | p<0.05 |
| *E. plumbeolata* | -0.0037 | -2.53 | p<0.05 |
| *E. pulchellata* | 0.0037 | 3.21 | p<0.05 |
| *E. pusillata* | -0.0040 | -3.98 | p<0.05 |
| *E. pygmaeata* | -0.0027 | -1.99 | p<0.05 |
| *E.satyrata* | -0.0063 | -5.81 | p<0.05 |
| *E. simpliciata* | -0.0022 | -1.85 | NS |
| *E.subfuscata* | -0.0009 | -0.82 | NS |
| *E.subumbrata* | -0.0037 | -2.58 | p<0.05 |
| *E. succenturiata* | -0.0077 | -6.47 | p<0.05 |
| *E. tantillaria* | -0.0006 | -0.63 | NS |
| *E. tenuiata* | 0.0007 | 0.82 | NS |
| *E. tripunctaria* | 0.0030 | 2.97 | p<0.05 |
| *E. trisignaria* | 0.0006 | 0.54 | NS |
| *E. valerianata* | -0.0030 | -2.10 | p<0.05 |
| *E. venosata* | -0.0062 | -6.54 | p<0.05 |
| *E. virgaureata* | 0.0051 | 4.72 | p<0.05 |
| *E. vulgata* | 0.0028 | 1.95 | NS |
| *E. quadripunctaria* | 0.0135 | 4.46 | p<0.05 |
| *Euplexia lucipara* | 0.0003 | 0.18 | NS |
| *Euproctis chrysorrhoea* | 0.0048 | 3.32 | p<0.05 |
| *E. similis* | -0.0040 | -1.87 | NS |

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|---|---|---|---|
| *Eupsilia transversa* | 0.0034 | 3.41 | p<0.05 |
| *Eurois occulta* | 0.0029 | 2.88 | p<0.05 |
| *Euthrix potatoria* | 0.0019 | 0.85 | NS |
| *Euxoa cursoria* | -0.0066 | -3.02 | p<0.05 |
| *E. nigricans* | -0.0124 | -14.20 | p<0.05 |
| *E. obelisca* | -0.0062 | -2.88 | p<0.05 |
| *E. tritici* | -0.0092 | -8.91 | p<0.05 |
| *Falcaria lacertinaria* | -0.0014 | -1.36 | NS |
| *Furcula bicuspis* | 0.0015 | 1.10 | NS |
| *F. bifida* | 0.0000 | -0.05 | NS |
| *F. furcula* | 0.0024 | 2.44 | p<0.05 |
| *Gastropacha quercifolia* | -0.0144 | -13.33 | p<0.05 |
| *Geometra papilionaria* | -0.0044 | -3.72 | p<0.05 |
| *Gnophos obfuscata* | -0.0140 | -5.03 | p<0.05 |
| *Gortyna flavago* | 0.0040 | 3.90 | p<0.05 |
| *Graphiphora augur* | -0.0184 | -18.62 | p<0.05 |
| *Gymnoscelis rufifasciata* | 0.0159 | 10.83 | p<0.05 |
| *Habrosyne pyritoides* | 0.0030 | 1.18 | NS |
| *Hada plebeja* | -0.0011 | -1.12 | NS |
| *Hadena albimacula* | -0.0081 | -1.83 | NS |
| *H. bicruris* | -0.0008 | -0.69 | NS |
| *H. compta* | 0.0004 | 0.29 | NS |
| *H. confusa* | -0.0055 | -5.65 | p<0.05 |
| *H. luteago* | -0.0001 | -0.03 | NS |
| *H. perplexa* | -0.0050 | -5.15 | p<0.05 |
| *H. rivularis* | -0.0020 | -2.17 | p<0.05 |
| *Hecatera bicolorata* | -0.0064 | -6.34 | p<0.05 |
| *Heliophobus reticulata* | -0.0070 | -6.99 | p<0.05 |
| *Heliothis maritima* | -0.0125 | -1.84 | NS |
| *H. viriplaca* | -0.0032 | -1.63 | NS |
| *Hemaris fuciformis* | -0.0025 | -1.88 | NS |
| *H. tityus* | 0.0015 | 1.25 | NS |
| *Hemistola chrysoprasaria* | 0.0032 | 2.45 | p<0.05 |
| *Hemithea aestivaria* | 0.0031 | 1.71 | NS |
| *Hepialus fusconebulosa* | -0.0008 | -0.58 | NS |
| *H. hecta* | -0.0070 | -8.37 | p<0.05 |
| *H. humuli* | -0.0058 | -3.99 | p<0.05 |
| *H. lupulinus* | 0.0020 | 1.21 | NS |
| *H. sylvina* | 0.0010 | 0.82 | NS |
| *Herminia grisealis* | 0.0035 | 2.64 | p<0.05 |
| *H. tarsicrinalis* | 0.0107 | 1.39 | NS |
| *Heterogenea asella* | 0.0037 | 0.82 | NS |
| *Hoplodrina alsines* | 0.0050 | 2.71 | p<0.05 |
| *H. ambigua* | 0.0118 | 7.43 | p<0.05 |
| *H. blanda* | 0.0037 | 2.58 | p<0.05 |
| *Horisme tersata* | -0.0047 | -3.14 | p<0.05 |
| *H. vitalbata* | 0.0018 | 1.13 | NS |
| *Hydraecia micacea* | 0.0029 | 1.77 | NS |
| *H. petasitis* | -0.0059 | -4.64 | p<0.05 |
| *Hydrelia flammeolaria* | 0.0040 | 3.86 | p<0.05 |
| *H. sylvata* | 0.0007 | 0.49 | NS |
| *Hydriomena furcata* | -0.0148 | -5.35 | p<0.05 |
| *H. impluviata* | 0.0003 | 0.38 | NS |
| *H. ruberata* | -0.0059 | -5.55 | p<0.05 |
| *Hylaea fasciaria* | -0.0026 | -2.54 | p<0.05 |
| *Hyles gallii* | -0.0027 | -2.10 | p<0.05 |
| *Hyloicus pinastri* | 0.0138 | 8.91 | p<0.05 |
| *Hypena crassalis* | 0.0011 | 0.68 | NS |
| *H. proboscidalis* | 0.0081 | 2.45 | p<0.05 |
| *Hypenodes humidalis* | 0.0005 | 0.32 | NS |
| *Hypomecis punctinalis* | -0.0005 | -0.33 | NS |
| *Hypomecis roboraria* | -0.0002 | -0.10 | NS |
| *Hyppa rectilinea* | -0.0006 | -0.28 | NS |
| *Idaea aversata* | 0.0093 | 2.75 | p<0.05 |
| *I. biselata* | 0.0089 | 4.33 | p<0.05 |
| *I. contiguaria* | -0.0049 | -1.23 | NS |
| *I. dilutaria* | -0.0098 | -2.08 | p<0.05 |
| *I. dimidiata* | 0.0065 | 3.41 | p<0.05 |
| *I. emarginata* | -0.0052 | -3.63 | p<0.05 |
| *I. fuscovenosa* | 0.0015 | 1.09 | NS |
| *I. muricata* | -0.0051 | -1.89 | NS |
| *I. rusticata* | 0.0159 | 7.36 | p<0.05 |
| *I. seriata* | 0.0047 | 4.36 | p<0.05 |
| *I. straminata* | -0.0023 | -2.37 | p<0.05 |
| *I. subsericeata* | -0.0038 | -3.10 | p<0.05 |
| *I. sylvestraria* | -0.0054 | -2.08 | p<0.05 |
| *I. trigeminata* | 0.0070 | 4.39 | p<0.05 |
| *Ipimorpha retusa* | -0.0009 | -0.64 | NS |
| *I. subtusa* | 0.0011 | 1.11 | NS |
| *Itame brunneata* | 0.0052 | 2.19 | p<0.05 |

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|---|---|---|---|
| *Jodia croceago* | -0.0061 | -3.71 | p<0.05 |
| *Jodis lactearia* | -0.0053 | -5.40 | p<0.05 |
| *Jordanita globulariae* | -0.0078 | -1.88 | NS |
| *Lacanobia contigua* | -0.0035 | -2.93 | p<0.05 |
| *L. oleracea* | 0.0025 | 1.02 | NS |
| *L. suasa* | -0.0010 | -0.83 | NS |
| *L. thalassina* | -0.0043 | -3.80 | p<0.05 |
| *L. w-latinum* | -0.0002 | -0.16 | NS |
| *Lampropteryx otregiata* | 0.0063 | 3.45 | p<0.05 |
| *L. suffumata* | 0.0020 | 1.82 | NS |
| *Laothoe populi* | 0.0087 | 3.18 | p<0.05 |
| *Larentia clavaria* | -0.0034 | -3.35 | p<0.05 |
| *Lasiocampa quercus* | -0.0065 | -5.54 | p<0.05 |
| *L. trifolii* | 0.0007 | 0.23 | NS |
| *Laspeyria flexula* | 0.0030 | 2.30 | p<0.05 |
| *Leucochlaena oditis* | -0.0108 | -1.36 | NS |
| *Leucoma salicis* | -0.0051 | -4.34 | p<0.05 |
| *Ligdia adustata* | 0.0018 | 1.31 | NS |
| *Lithomoia solidaginis* | -0.0080 | -6.21 | p<0.05 |
| *Lithophane hepatica* | 0.0126 | 13.13 | p<0.05 |
| *L. leautieri* | 0.0108 | 8.76 | p<0.05 |
| *L. ornitopus* | 0.0082 | 7.61 | p<0.05 |
| *L. semibrunnea* | 0.0019 | 1.85 | NS |
| *Lithosia quadra* | 0.0102 | 7.52 | p<0.05 |
| *Lithostege griseata* | 0.0080 | 1.84 | NS |
| *Lobophora halterata* | -0.0008 | -0.78 | NS |
| *Lomaspilis marginata* | 0.0013 | 0.51 | NS |
| *Lomographa bimaculata* | 0.0039 | 3.48 | p<0.05 |
| *L. temerata* | 0.0085 | 4.78 | p<0.05 |
| *Luperina nickerlii* | 0.0020 | 0.37 | NS |
| *L. testacea* | -0.0030 | -1.76 | NS |
| *Lycia hirtaria* | -0.0031 | -2.69 | p<0.05 |
| *L. lapponaria* | -0.0044 | -1.24 | NS |
| *Lycia zonaria* | -0.0056 | -0.95 | NS |
| *Lycophotia porphyrea* | -0.0076 | -4.84 | p<0.05 |
| *Lygephila pastinum* | -0.0005 | -0.49 | NS |
| *Lymantria dispar* | -0.0014 | -0.35 | NS |
| *L. monacha* | 0.0129 | 8.44 | p<0.05 |
| *Macaria alternata* | 0.0081 | 3.98 | p<0.05 |

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|---|---|---|---|
| *M. carbonaria* | -0.0040 | -0.66 | NS |
| *M. liturata* | -0.0031 | -3.10 | p<0.05 |
| *M. notata* | 0.0006 | 0.46 | NS |
| *M. wauaria* | -0.0136 | -17.65 | p<0.05 |
| *Macrochilo cribrumalis* | 0.0005 | 0.20 | NS |
| *Macroglossum stellatarum* | 0.0183 | 14.01 | p<0.05 |
| *Macrothylacia rubi* | -0.0061 | -5.16 | p<0.05 |
| *Malacosoma castrensis* | -0.0089 | -1.48 | NS |
| *M. neustria* | -0.0187 | -10.84 | p<0.05 |
| *Mamestra brassicae* | -0.0096 | -7.77 | p<0.05 |
| *Meganola albula* | 0.0076 | 3.10 | p<0.05 |
| *Meganola strigula* | -0.0056 | -1.83 | NS |
| *Melanchra persicariae* | -0.0048 | -2.84 | p<0.05 |
| *M. pisi* | -0.0142 | -11.61 | p<0.05 |
| *Melanthia procellata* | -0.0046 | -3.38 | p<0.05 |
| *Menophra abruptaria* | 0.0045 | 3.36 | p<0.05 |
| *Mesoleuca albicillata* | -0.0050 | -5.99 | p<0.05 |
| *Mesoligia furuncula* | 0.0009 | 0.75 | NS |
| *M. literosa* | -0.0045 | -4.18 | p<0.05 |
| *Miltochrista miniata* | 0.0074 | 4.41 | p<0.05 |
| *Mimas tiliae* | 0.0071 | 4.96 | p<0.05 |
| *Minoa murinata* | -0.0049 | -1.72 | NS |
| *Moma alpium* | 0.0050 | 1.41 | NS |
| *Mormo maura* | 0.0018 | 1.96 | NS |
| *Mythimna albipuncta* | 0.0192 | 10.96 | p<0.05 |
| *M. comma* | -0.0040 | -3.34 | p<0.05 |
| *M. conigera* | -0.0068 | -6.31 | p<0.05 |
| *M. favicolor* | -0.0018 | -0.45 | NS |
| *M. ferrago* | -0.0044 | -2.38 | p<0.05 |
| *M. flammea* | -0.0074 | -2.22 | p<0.05 |
| *M. impura* | -0.0021 | -0.61 | NS |
| *M. l-album* | 0.0117 | 4.42 | p<0.05 |
| *M. litoralis* | -0.0053 | -2.22 | p<0.05 |
| *M. obsoleta* | -0.0043 | -2.60 | p<0.05 |
| *M. pallens* | -0.0002 | -0.10 | NS |
| *M. pudorina* | -0.0008 | -0.63 | NS |
| *M. putrescens* | -0.0037 | -0.89 | NS |
| *M. straminea* | 0.0008 | 0.67 | NS |

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|---|---|---|---|
| *M. turca* | 0.0013 | 0.66 | NS |
| *M. unipuncta* | 0.0073 | 5.00 | p<0.05 |
| *Naenia typica* | -0.0034 | -3.74 | p<0.05 |
| *Nebula salicata* | -0.0093 | -7.06 | p<0.05 |
| *Noctua comes* | 0.0035 | 1.14 | NS |
| *N. fimbriata* | 0.0074 | 5.43 | p<0.05 |
| *N. interjecta* | 0.0030 | 2.05 | p<0.05 |
| *N. janthe* | 0.0053 | 1.92 | NS |
| *N. orbona* | -0.0058 | -4.78 | p<0.05 |
| *N. pronuba* | 0.0198 | 2.20 | p<0.05 |
| *Nola confusalis* | 0.0063 | 6.31 | p<0.05 |
| *N. cucullatella* | -0.0060 | -4.34 | p<0.05 |
| *Nonagria typhae* | 0.0007 | 0.71 | NS |
| *Notodonta dromedarius* | 0.0064 | 4.23 | p<0.05 |
| *N. ziczac* | 0.0060 | 3.77 | p<0.05 |
| *Nudaria mundana* | -0.0041 | -3.47 | p<0.05 |
| *Nycteola revayana* | 0.0068 | 7.00 | p<0.05 |
| *Ochropacha duplaris* | 0.0000 | 0.00 | NS |
| *Ochropleura plecta* | 0.0161 | 3.94 | p<0.05 |
| *Odezia atrata* | -0.0085 | -6.43 | p<0.05 |
| *Odontopera bidentata* | -0.0010 | -0.69 | NS |
| *Odontosia carmelita* | -0.0019 | -1.34 | NS |
| *Oligia fasciuncula* | -0.0007 | -0.39 | NS |
| *O. latruncula* | -0.0043 | -4.16 | p<0.05 |
| *O. strigilis* | -0.0067 | -6.22 | p<0.05 |
| *O. versicolor* | 0.0000 | -0.05 | NS |
| *Omphaloscelis lunosa* | 0.0044 | 3.47 | p<0.05 |
| *Operophtera brumata* | -0.0071 | -6.14 | p<0.05 |
| *O. fagata* | -0.0066 | -7.00 | p<0.05 |
| *Opisthograptis luteolata* | 0.0085 | 1.76 | NS |
| *Orgyia antiqua* | 0.0002 | 0.23 | NS |
| *O. recens* | -0.0054 | -1.67 | NS |
| *Oria musculosa* | -0.0143 | -5.48 | p<0.05 |
| *Orthonama vittata* | -0.0027 | -2.92 | p<0.05 |
| *Orthosia cerasi* | 0.0100 | 5.48 | p<0.05 |
| *O. cruda* | 0.0049 | 3.92 | p<0.05 |
| *O. gothica* | 0.0103 | 4.55 | p<0.05 |
| *O. gracilis* | -0.0013 | -1.31 | NS |
| *O. incerta* | 0.0069 | 4.35 | p<0.05 |
| *O. miniosa* | -0.0040 | -3.55 | p<0.05 |

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|---|---|---|---|
| *O. munda* | 0.0063 | 5.66 | p<0.05 |
| *O. opima* | -0.0047 | -4.37 | p<0.05 |
| *O. populeti* | -0.0021 | -2.01 | p<0.05 |
| *Ourapteryx sambucaria* | -0.0019 | -0.96 | NS |
| *Pachycnemia hippocastanaria* | -0.0010 | -0.38 | NS |
| *Panemeria tenebrata* | -0.0048 | -5.27 | p<0.05 |
| *Panolis flammea* | 0.0055 | 6.29 | p<0.05 |
| *Papestra biren* | -0.0031 | -3.06 | p<0.05 |
| *Paracolax tristalis* | -0.0098 | -2.58 | p<0.05 |
| *Paradarisa consonaria* | 0.0021 | 1.64 | NS |
| *Paradrina clavipalpis* | -0.0032 | -2.91 | p<0.05 |
| *Parascotia fuliginaria* | 0.0047 | 2.35 | p<0.05 |
| *Parasemia plantaginis* | -0.0070 | -7.06 | p<0.05 |
| *Parastichtis suspecta* | -0.0032 | -3.27 | p<0.05 |
| *P. ypsillon* | -0.0011 | -1.16 | NS |
| *Parectropis similaria* | 0.0019 | 1.28 | NS |
| *Pasiphila chloerata* | -0.0008 | -0.53 | NS |
| *P. debiliata* | 0.0004 | 0.19 | NS |
| *P. rectangulata* | 0.0056 | 4.16 | p<0.05 |
| *Pechipogo strigilata* | -0.0081 | -6.66 | p<0.05 |
| *Pelosia muscerda* | 0.0074 | 1.81 | NS |
| *Pelurga comitata* | -0.0077 | -7.75 | p<0.05 |
| *Perconia strigillaria* | -0.0034 | -2.06 | p<0.05 |
| *Peribatodes rhomboidaria* | 0.0017 | 0.79 | NS |
| *Peridea anceps* | -0.0001 | -0.08 | NS |
| *Perizoma affinitata* | -0.0007 | -0.73 | NS |
| *P. albulata* | -0.0054 | -6.45 | p<0.05 |
| *P. alchemillata* | -0.0027 | -1.93 | NS |
| *P. bifaciata* | -0.0030 | -2.92 | p<0.05 |
| *P. blandiata* | -0.0054 | -3.23 | p<0.05 |
| *P. didymata* | -0.0147 | -11.78 | p<0.05 |
| *P. flavofasciata* | -0.0010 | -0.96 | NS |
| *P. minorata* | -0.0089 | -4.30 | p<0.05 |
| *P. sagittata* | -0.0124 | -2.78 | p<0.05 |
| *P. taeniata* | 0.0000 | 0.00 | NS |
| *Petrophora chlorosata* | 0.0038 | 2.31 | p<0.05 |
| *Phalera bucephala* | 0.0006 | 0.35 | NS |

| | | | |
|---|---|---|---|
| *Pheosia gnoma* | 0.0028 | 1.82 | NS |
| *P. tremula* | 0.0055 | 4.18 | p<0.05 |
| *Phibalapteryx virgata* | -0.0059 | -2.81 | p<0.05 |
| *Phigalia pilosaria* | -0.0005 | -0.52 | NS |
| *Philereme transversata* | 0.0000 | 0.03 | NS |
| *P. vetulata* | -0.0035 | -2.41 | p<0.05 |
| *Phlogophora meticulosa* | 0.0111 | 4.30 | p<0.05 |
| *Photedes captiuncula* | -0.0079 | -2.57 | p<0.05 |
| *P. minima* | -0.0058 | -5.22 | p<0.05 |
| *Phragmatobia fuliginosa* | 0.0061 | 3.67 | p<0.05 |
| *Phytometra viridaria* | -0.0049 | -5.00 | p<0.05 |
| *Plagodis dolabraria* | 0.0069 | 5.26 | p<0.05 |
| *P. pulveraria* | -0.0036 | -3.70 | p<0.05 |
| *Plemyria rubiginata* | -0.0006 | -0.68 | NS |
| *Plusia festucae* | 0.0060 | 5.59 | p<0.05 |
| *P. putnami* | 0.0061 | 4.12 | p<0.05 |
| *Poecilocampa populi* | -0.0022 | -2.49 | p<0.05 |
| *Polia bombycina* | -0.0120 | -11.35 | p<0.05 |
| *P. nebulosa* | -0.0037 | -3.69 | p<0.05 |
| *P. trimaculosa* | -0.0088 | -6.32 | p<0.05 |
| *Polychrysia moneta* | -0.0094 | -10.51 | p<0.05 |
| *Polymixis flavicincta* | 0.0022 | 1.77 | NS |
| *P. lichenea* | -0.0017 | -1.00 | NS |
| *P. xanthomista* | -0.0054 | -1.23 | NS |
| *Polyploca ridens* | 0.0020 | 1.80 | NS |
| *Protodeltote pygarga* | 0.0130 | 9.88 | p<0.05 |
| *Protolampra sobrina* | -0.0069 | -1.59 | NS |
| *Pseudoips prasinana* | 0.0048 | 4.45 | p<0.05 |
| *Pseudopanthera macularia* | -0.0053 | -4.58 | p<0.05 |
| *Pseudoterpna pruinata* | -0.0094 | -9.10 | p<0.05 |
| *Pterapherapteryx sexalata* | -0.0004 | -0.36 | NS |
| *Pterostoma palpina* | 0.0044 | 2.95 | p<0.05 |
| *Ptilodon capucina* | -0.0014 | -0.72 | NS |
| *P. cucullina* | 0.0069 | 2.87 | p<0.05 |
| *Ptilophora plumigera* | -0.0045 | -1.27 | NS |

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|---|---|---|---|
| *Pyrrhia umbra* | -0.0032 | -2.95 | p<0.05 |
| *Rheumaptera cervinalis* | -0.0037 | -2.84 | p<0.05 |
| *R. hastata* | -0.0040 | -3.55 | p<0.05 |
| *R. undulata* | -0.0001 | -0.15 | NS |
| *Rhizedra lutosa* | 0.0012 | 1.31 | NS |
| *Rhyacia simulans* | -0.0163 | -13.56 | p<0.05 |
| *Rivula sericealis* | 0.0333 | 16.09 | p<0.05 |
| *Rusina ferruginea* | -0.0056 | -4.01 | p<0.05 |
| *Saturnia pavonia* | -0.0079 | -7.04 | p<0.05 |
| *Schrankia costaestrigalis* | 0.0085 | 8.96 | p<0.05 |
| *S. taenialis* | -0.0023 | -1.19 | NS |
| *Scoliopteryx libatrix* | -0.0018 | -1.38 | NS |
| *Scopula emutaria* | -0.0020 | -0.65 | NS |
| *S. floslactata* | -0.0010 | -1.10 | NS |
| *S. imitaria* | 0.0032 | 2.09 | p<0.05 |
| *S. immutata* | -0.0008 | -0.78 | NS |
| *S. marginepunctata* | -0.0003 | -0.19 | NS |
| *S. ornata* | -0.0022 | -0.88 | NS |
| *S. rubiginata* | 0.0034 | 0.93 | NS |
| *S. ternata* | -0.0025 | -1.84 | NS |
| *Scotopteryx bipunctaria* | -0.0057 | -3.45 | p<0.05 |
| *S. chenopodiata* | -0.0119 | -6.49 | p<0.05 |
| *S. luridata* | -0.0082 | -8.30 | p<0.05 |
| *S. mucronata* | -0.0050 | -4.36 | p<0.05 |
| *Selenia dentaria* | -0.0002 | -0.09 | NS |
| *S. lunularia* | -0.0076 | -8.65 | p<0.05 |
| *S. tetralunaria* | 0.0033 | 2.67 | p<0.05 |
| *Selidosema brunnearia* | -0.0098 | -3.07 | p<0.05 |
| *Semiaspilates ochrearia* | 0.0028 | 1.41 | NS |
| *Sesia bembeciformis* | -0.0037 | -3.55 | p<0.05 |
| *Setina irrorella* | -0.0081 | -2.11 | p<0.05 |
| *Shargacucullia lychnitis* | -0.0007 | -0.18 | NS |
| *S. verbasci* | -0.0031 | -2.86 | p<0.05 |
| *Sideridis albicolon* | -0.0042 | -2.09 | p<0.05 |
| *Simyra albovenosa* | 0.0057 | 2.11 | p<0.05 |
| *Smerinthus ocellata* | 0.0001 | 0.12 | NS |
| *Spaelotis ravida* | -0.0175 | -13.65 | p<0.05 |
| *Spargania luctuata* | -0.0037 | -0.94 | NS |

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|---|---|---|---|
| *Sphinx ligustri* | 0.0011 | 0.81 | NS |
| *Spilosoma lubricipeda* | 0.0010 | 0.38 | NS |
| *S. luteum* | 0.0059 | 2.40 | p<0.05 |
| *S. urticae* | -0.0021 | -1.05 | NS |
| *Standfussiana lucernea* | -0.0056 | -3.63 | p<0.05 |
| *Stauropus fagi* | 0.0045 | 3.16 | p<0.05 |
| *Stilbia anomala* | -0.0091 | -7.37 | p<0.05 |
| *Syngrapha interrogationis* | -0.0016 | -1.29 | NS |
| *Tethea ocularis* | 0.0040 | 3.05 | p<0.05 |
| *T. or* | -0.0019 | -1.57 | NS |
| *Tetheella fluctuosa* | 0.0015 | 0.73 | NS |
| *Thalpophila matura* | -0.0077 | -6.12 | p<0.05 |
| *Thera britannica* | 0.0152 | 13.16 | p<0.05 |
| *T. cognata* | -0.0034 | -1.92 | NS |
| *T. cupressata* | 0.0248 | 8.24 | p<0.05 |
| *T. firmata* | 0.0033 | 3.43 | p<0.05 |
| *T. juniperata* | 0.0001 | 0.08 | NS |
| *T. obeliscata* | -0.0014 | -1.07 | NS |
| *T. primaria* | -0.0074 | -8.38 | p<0.05 |
| *Tholera cespitis* | -0.0070 | -8.37 | p<0.05 |
| *T. decimalis* | -0.0081 | -8.20 | p<0.05 |
| *Thumatha senex* | -0.0013 | -1.09 | NS |
| *Thyatira batis* | 0.0002 | 0.14 | NS |
| *Timandra comae* | 0.0063 | 3.26 | p<0.05 |
| *Trichiura crataegi* | -0.0087 | -8.84 | p<0.05 |
| *Trichopteryx carpinata* | 0.0032 | 3.27 | p<0.05 |
| *T. polycommata* | -0.0043 | -1.96 | NS |
| *Triphosa dubitata* | -0.0049 | -6.07 | p<0.05 |
| *Trisateles emortualis* | 0.0115 | 2.27 | p<0.05 |
| *Tyria jacobaeae* | 0.0044 | 2.14 | p<0.05 |
| *Tyta luctuosa* | -0.0011 | -0.85 | NS |
| *Venusia cambrica* | -0.0020 | -1.56 | NS |

| | | | |
|---|---|---|---|
| *Watsonalla binaria* | 0.0033 | 2.40 | p<0.05 |
| *W. cultraria* | 0.0001 | 0.12 | NS |
| *Xanthia aurago* | 0.0069 | 5.64 | p<0.05 |
| *X. citrago* | 0.0022 | 2.41 | p<0.05 |
| *X. gilvago* | -0.0051 | -5.90 | p<0.05 |
| *X. icteritia* | -0.0034 | -2.94 | p<0.05 |
| *X. ocellaris* | -0.0068 | -2.49 | p<0.05 |
| *X. togata* | 0.0005 | 0.54 | NS |
| *Xanthorhoe biriviata* | 0.0029 | 1.03 | NS |
| *X. decoloraria* | -0.0100 | -6.06 | p<0.05 |
| *X. designata* | 0.0123 | 7.97 | p<0.05 |
| *X. ferrugata* | -0.0108 | -8.47 | p<0.05 |
| *X. fluctuata* | -0.0060 | -2.41 | p<0.05 |
| *X. montanata* | -0.0134 | -2.48 | p<0.05 |
| *X. quadrifasiata* | 0.0046 | 3.01 | p<0.05 |
| *X. spadicearia* | -0.0001 | -0.08 | NS |
| *Xestia agathina* | 0.0000 | 0.00 | NS |
| *X. alpicola* | -0.0021 | -0.75 | NS |
| *X. ashworthii* | -0.0096 | -1.89 | NS |
| *X. baja* | -0.0085 | -7.17 | p<0.05 |
| *X. castanea* | -0.0032 | -3.15 | p<0.05 |
| *X. c-nigrum* | 0.0062 | 3.28 | p<0.05 |
| *X. ditrapezium* | -0.0024 | -2.22 | p<0.05 |
| *X. rhomboidea* | 0.0011 | 0.89 | NS |
| *X. sexstrigata* | -0.0023 | -1.88 | NS |
| *X. triangulum* | 0.0041 | 2.14 | p<0.05 |
| *X. xanthographa* | 0.0040 | 1.56 | NS |
| *Xylena exsoleta* | -0.0045 | -5.59 | p<0.05 |
| *X. vetusta* | 0.0050 | 5.53 | p<0.05 |
| *Xylocampa areola* | 0.0062 | 4.70 | p<0.05 |
| *Zanclognatha tarsipennalis* | 0.0104 | 7.05 | p<0.05 |
| *Zeuzera pyrina* | 0.0020 | 1.53 | NS |
| *Zygaena filipendulae* | -0.0018 | -1.45 | NS |
| *Z. lonicerae* | -0.0035 | -2.86 | p<0.05 |
| *Z.purpuralis* | -0.0164 | -1.43 | NS |
| *Zygaena trifolii* | -0.0097 | -8.67 | p<0.05 |

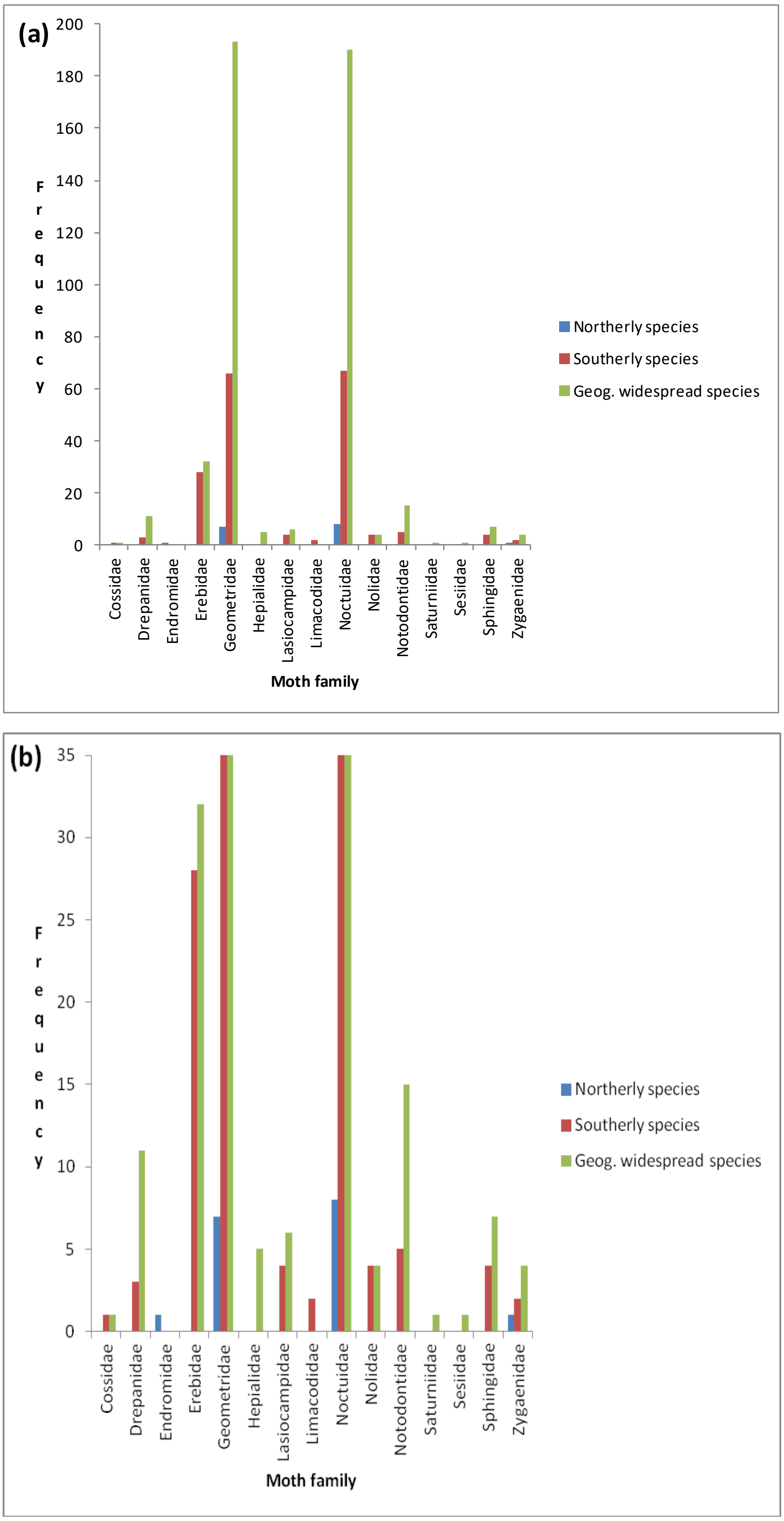


**Fig. A3.1** The frequency (a), and proportion (b), of moths in different distribution groupings (northerly, southerly, geographically widespread) by taxonomic family.

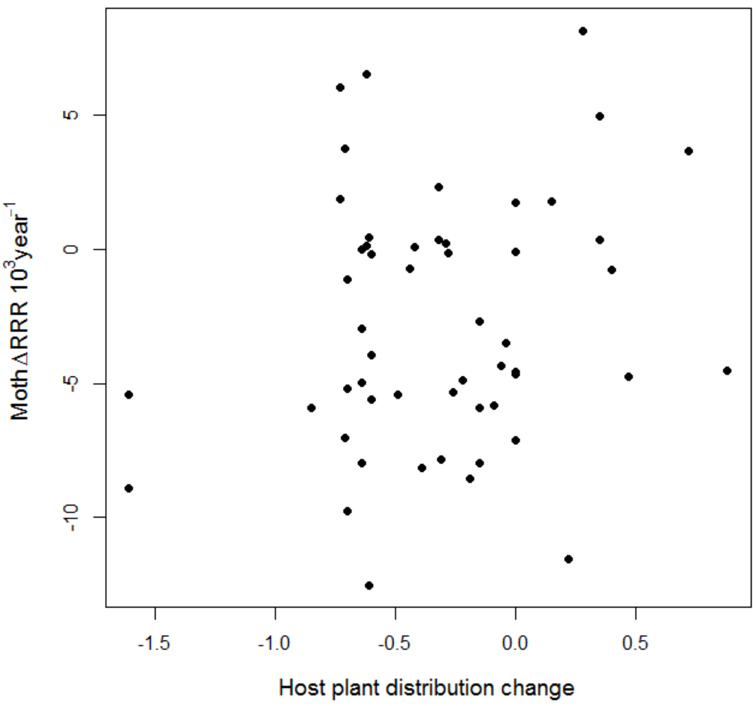


**Fig. A3.2** Change in the frequency of occurrence of monophagous moth species 1970-99 versus 2000-10 in relation to change in host plant distribution between 1970 and 1987 (with outliers removed as per main text).

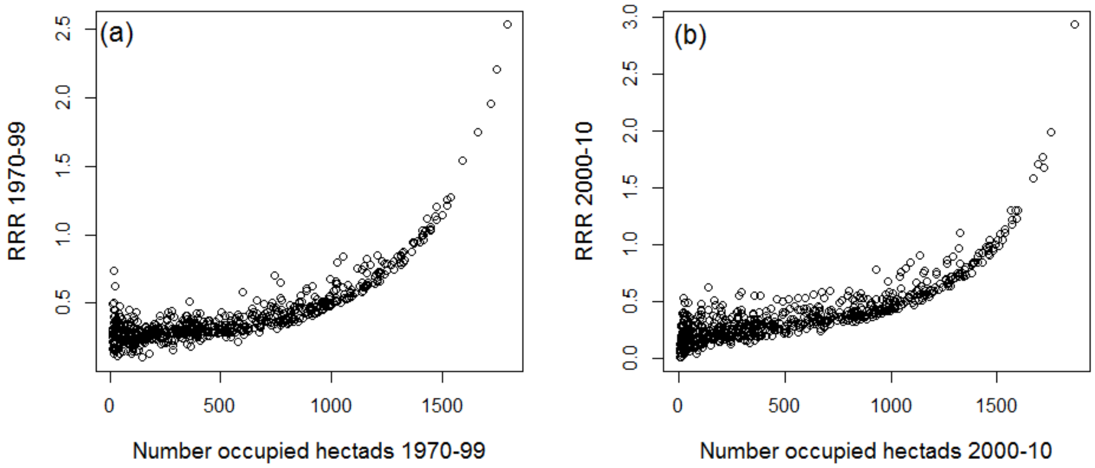


**Fig. A3.3** Relationship between number of hectads (grid squares) occupied by each species in (a) 1970-99 and (b) 2000-10 versus the relative reporting rate (RRR) which indicates frequency of occurrence after standardising for recorder effort. The curved relationships show that the most common species is approximately 2.5 times more common than the average benchmark species, whilst rare species are only approximately 0.25 times as common as the average benchmark species.

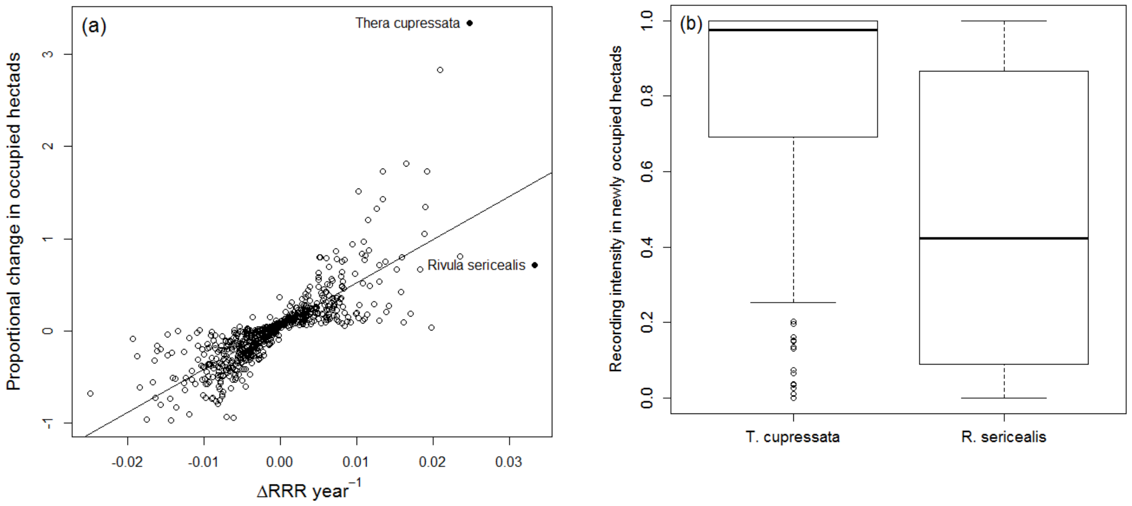


**Fig. A3.4** Change in moth frequency of occurrence versus proportional change in hectads. Panel (a) shows the relationship between change in moth frequency of occurrence (ΔRRR) using the Frescalo method which controls for spatiotemporal variation in recorder effort versus proportional change in hectads (grid squares) listed as occupied. The time periods under consideration are 1970-99 versus 2000-10. The correlation is significant ($F_{1,161} = 1301$, $p < 0.001$). We select two outliers from this relationship, both with large positive RRR trends, to demonstrate how the Frescalo method accounts for variation in recorder effort (shown in panel (b)). Panel (a) suggests that the species *T. cupressata* has a greater increase in records than expected from changes in frequency relative to benchmark species (which inform on recorder effort). Hence, the large increase in number of records is partly due to increased recording effort in these locations in the latter period. In contrast, *R. sericealis* has fewer records than expected which indicates relatively less recording effort focussed in areas where this species occurs. Panel (b) shows the recording intensity in hectads newly occupied by the two moths in the latter period. Recording intensity is calculated as the proportion of benchmarks species recorded in a hectad. Hence, this confirms that *T. cupressata* has expanded into well recorded areas, whilst *R. sericealis* has expanded into poorly recorded areas.

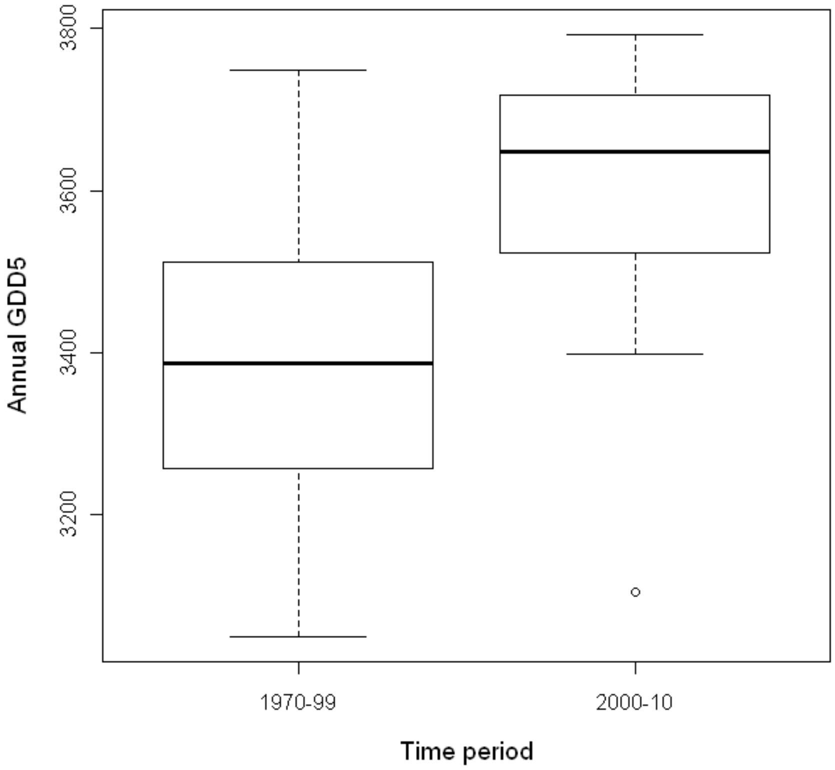


**Fig. A3.5** Annual accumulated temperatures (growing degree days > 5°C; GDD5; calculated from daily mean temperature data from the UK Met Office Central England Temperature dataset; http://www.metoffice.gov.uk/hadobs/hadcet/) during the two recording periods. The box and whisker plots show median annual GDD5, upper and lower quartiles and 95th percentiles. Annual GDD5 values were significantly higher in the latter period (t-test: t = 2.84, df = 17.71, *p* = 0.01).

# Appendix 4: Supporting Information for Chapter 4: Using citizen science butterfly counts to predict species population trends

**Table A4.1** Official Big Butterfly Count period for each year. The period changes each year to include weekends in order to maximise participation.

| Year | Period |
|---|---|
| 2010 | 24th July – 1st August |
| 2011 | 16th July – 7th August |
| 2012 | 14th July – 5th August |
| 2013 | 20th July – 11th August |
| 2014 | 19th July – 10th August |
| 2015 | 17th July – 9th August |

**Table A4.2** Common UK butterflies counted by the Big Butterfly Count (BBC) and considered in Chapter 4. Voltinism refers to the assumed species' seasonal pattern, where U and B represent univoltine and bivoltine species, respectively, and S represents species with more than two broods or a complicated seasonal pattern (where a spline rather than phenomenological generalised abundance index (GAI) was fitted to the UKBMS data). nSq is the number of 1km squares in which the species was recorded by the BBC in the period 2010-2015.

| Common name | Species | Species | Voltinism | nSq |
|---|---|---|---|---|
| Marbled White | *Melanargia galathea* | MW | U | 7818 |
| Large Skipper | *Ochlodes sylvanus* | LS | U | 12226 |
| Ringlet | *Aphantopus* | R | U | 25351 |
| Meadow Brown | *Maniola jurtina* | MB | U | 49277 |
| Gatekeeper | *Pyronia tithonus* | G | U | 65175 |
| Brimstone | *Gonepteryx rhamni* | B | B | 13673 |
| Holly Blue | *Celastrina argiolus* | HB | B | 15120 |
| Common Blue | *Polyommatus icarus* | CB | B | 24033 |
| Green-veined White | *Pieris napi* | GvW | B | 28187 |
| Large White | *Pieris brassicae* | LW | B | 75801 |
| Small White | *Pieris rapae* | SmW | B | 81553 |
| Small Copper | *Lycaena phlaeas* | SC | S | 11584 |
| Painted Lady | *Vanessa cardui* | PL | S | 13867 |
| Speckled Wood | *Pararge aegeria* | SpW | S | 28010 |
| Comma | *Polygonia c-album* | C | S | 34952 |
| Red Admiral | *Vanessa atalanta* | RA | S | 45018 |
| Small Tortoiseshell | *Aglais urticae* | ST | S | 47904 |
| Peacock | *Aglais io* | P | S | 56907 |

**Table A4.3** Summary of Big Butterfly Count data 2011-2015.

| Year | 2011 | 2012 | 2013 | 2014 | 2015 |
|---|---|---|---|---|---|
| Number of 15 minute counts | 28715 | 24074 | 44108 | 42768 | 49090 |
| Mean number of counts per location | 1.31 | 1.35 | 1.48 | 1.50 | 1.23 |
| Mean number of species seen per count | 4.00 | 3.24 | 4.52 | 4.12 | 4.01 |

**Table A4.4** Percentage of Big Butterfly Count counts recorded in different habitat categories each year.

| Location | 2011 | 2012 | 2013 | 2014 | 2015 |
|---|---|---|---|---|---|
| Field | 13.2 | 18.1 | 10.2 | 9.1 | 11.7 |
| Garden | 65.1 | 54.7 | 67.3 | 70.1 | 65.8 |
| Other rural | 10.5 | 14.6 | 11.0 | 10.1 | 11.1 |
| Other urban | 2.4 | 3.5 | 4.5 | 3.5 | 3.5 |
| Park | 3.3 | 3.9 | 3.4 | 3.6 | 3.6 |
| School | 0.8 | 0.2 | 0.4 | 0.3 | 0.3 |
| Wood | 4.6 | 5.0 | 3.3 | 3.3 | 4.0 |

**Table A4.5** Mean and standard error (SE) of the percentage habitat types (LCM2007) for 1km squares sampled by the UK Butterfly Monitoring Scheme (UKBMS) and Big Butterfly Count (BBC) 2010-2014.

| Habitat type | UKBMS | | BBC | |
|---|---|---|---|---|
| | Mean | SE | Mean | SE |
| Broadleaf woodland | 13.77 | 0.34 | 7.76 | 0.04 |
| Coniferous woodland | 4.39 | 0.25 | 1.53 | 0.02 |
| Arable and Horticulture | 27.55 | 0.54 | 23.19 | 0.08 |
| Improved Grassland | 26.38 | 0.42 | 24.37 | 0.06 |
| Semi-natural grassland | 7.90 | 0.23 | 4.85 | 0.03 |
| Mountain, heath & bog | 3.92 | 0.25 | 1.20 | 0.02 |
| Saltwater | 0.36 | 0.06 | 0.37 | 0.01 |
| Freshwater | 1.05 | 0.09 | 0.86 | 0.01 |
| Coastal | 1.62 | 0.17 | 1.09 | 0.02 |
| Urban & Suburban | 9.80 | 0.38 | 33.36 | 0.11 |

**Table A4.6** Estimated net population change 2011-2014 (log growth rate) from Big Butterfly Count (BBC) and the UK Butterfly Monitoring Scheme (UKBMS), with 95% confidence intervals. Asterisks indicate significant change ($p < 0.05$).

| Common name | BBC | | UKBMS | | Difference | |
|---|---|---|---|---|---|---|
| Brimstone | 0.44 (0.13, 0.76) | * | 0.50 (0.41, 0.58) | * | -0.05 (-0.38, 0.27) | |
| Comma | 0.29 (0.14, 0.44) | * | -0.04 (-0.14, 0.06) | | 0.33 (0.15, 0.51) | * |
| Common Blue | 0.24 (0.03, 0.45) | * | 1.14 (1.05, 1.23) | * | -0.90 (-1.13, -0.67) | * |
| Gatekeeper | 0.23 (0.14, 0.32) | * | 0.25 (0.21, 0.30) | * | -0.02 (-0.12, 0.08) | |
| Green-veined White | 0.15 (0, 0.29) | | 0.10 (0.04, 0.17) | * | 0.04 (-0.12, 0.2) | |
| Holly Blue | -0.29 (-0.52, -0.05) | * | -0.72 (-0.83, -0.61) | * | 0.44 (0.18, 0.70) | * |
| Large Skipper | 0.98 (0.68, 1.27) | * | 0.50 (0.39, 0.62) | * | 0.48 (0.16, 0.80) | * |
| Large White | -0.04 (-0.14, 0.06) | | -0.24 (-0.32, -0.16) | * | 0.20 (0.07, 0.33) | * |
| Marbled White | 0.56 (0.22, 0.90) | * | 0.84 (0.72, 0.96) | * | -0.28 (-0.65, 0.08) | |
| Meadow Brown | 0.60 (0.50, 0.71) | * | 0.40 (0.35, 0.45) | * | 0.20 (0.09, 0.32) | * |
| Painted Lady | 0.44 (0.11, 0.77) | * | 0.61 (0.42, 0.79) | * | -0.17 (-0.55, 0.22) | |
| Peacock | 1.16 (1.02, 1.31) | * | 1.06 (0.98, 1.15) | * | 0.10 (-0.07, 0.27) | |
| Red Admiral | -0.30 (-0.42, -0.17) | * | -0.77 (-0.85, -0.69) | * | 0.47 (0.32, 0.62) | * |
| Ringlet | 0.58 (0.41, 0.76) | * | 0.17 (0.09, 0.26) | * | 0.41 (0.21, 0.61) | * |
| Small Copper | -0.10 (-0.36, 0.15) | | -0.43 (-0.53, -0.33) | * | 0.32 (0.05, 0.60) | * |
| Small Tortoiseshell | 1.14 (1, 1.28) | * | 1.79 (1.65, 1.92) | * | -0.65 (-0.84, -0.45) | * |
| Small White | 0.03 (-0.06, 0.11) | | -0.13 (-0.21, -0.05) | * | 0.15 (0.04, 0.27) | * |
| Speckled Wood | -0.34 (-0.49, -0.19) | * | -0.26 (-0.31, -0.20) | * | -0.08 (-0.25, 0.08) | |

**Table A4.7** Overdispersion estimated by the ratio of the Pearson Chi-squared statistic to its degrees of freedom. UKBMS = UK Butterfly Monitoring Scheme, BBC = Big Butterfly Count.

| Species | UKBMS | BBC |
|---|---:|---:|
| Brimstone | 2.48 | 1.40 |
| Comma | 1.55 | 1.35 |
| Common Blue | 10.14 | 2.26 |
| Gatekeeper | 14.01 | 3.23 |
| Green-veined White | 7.27 | 2.38 |
| Holly Blue | 0.89 | 1.41 |
| Large Skipper | 3.27 | 2.65 |
| Large White | 6.18 | 2.95 |
| Marbled White | 8.36 | 3.63 |
| Meadow Brown | 27.33 | 4.06 |
| Painted Lady | 0.74 | 1.81 |
| Peacock | 6.80 | 4.06 |
| Red Admiral | 2.00 | 1.65 |
| Ringlet | 11.87 | 3.07 |
| Small Copper | 2.30 | 1.78 |
| Small Tortoiseshell | 3.18 | 2.79 |
| Small White | 7.99 | 2.68 |
| Speckled Wood | 3.59 | 1.64 |

**Table A4.8** Residual deviance, degrees of freedom (df) and associated ratios from fitting a Poisson and negative-binomial GLM where the response is the total Big Butterfly Count count per day and measures of effort (log number of counts made) and phenology (from the corresponding GAI curve) as covariates.

| | | Residual deviance | | Ratio | |
|---|---|---|---|---|---|
| Species | df | Poisson | NegBin | Poisson | NegBin |
| Brimstone | 216 | 2362.6 | 253.5 | 10.9 | 1.2 |
| Comma | 230 | 1602.4 | 282.7 | 7.0 | 1.2 |
| Common Blue | 230 | 7362 | 261.3 | 32.0 | 1.1 |
| Gatekeeper | 230 | 11380 | 283.7 | 49.5 | 1.2 |
| Green-veined White | 222 | 4452.6 | 285 | 20.1 | 1.3 |
| Holly Blue | 215 | 1559.3 | 274.3 | 7.3 | 1.3 |
| Large Skipper | 207 | 3019.7 | 240.6 | 14.6 | 1.2 |
| Large White | 232 | 9695.7 | 263.9 | 41.8 | 1.1 |
| Marbled White | 203 | 5598.1 | 230.9 | 27.6 | 1.1 |
| Meadow Brown | 234 | 31229.3 | 266 | 133.5 | 1.1 |
| Painted Lady | 223 | 2306.8 | 256.3 | 10.3 | 1.1 |
| Peacock | 229 | 35926.9 | 258 | 156.9 | 1.1 |
| Red Admiral | 233 | 5711 | 270.7 | 24.5 | 1.2 |
| Ringlet | 229 | 9034.6 | 264.1 | 39.5 | 1.2 |
| Small Copper | 220 | 2103.3 | 264.6 | 9.6 | 1.2 |
| Small Tortoiseshell | 234 | 13983.2 | 255.5 | 59.8 | 1.1 |
| Small White | 235 | 5484.7 | 327.5 | 23.3 | 1.4 |
| Speckled Wood | 233 | 2782.7 | 279 | 11.9 | 1.2 |

**Table A4.9** Relative importance (Grömping 2006) of Big Butterfly Count (BBC) and weather for each species. The same conclusions were obtained by considering all possible regressions based on AIC.

| Species | BBC | Weather |
|---|---:|---:|
| Small Tortoiseshell | 0.99 | 0.01 |
| Large White | 0.88 | 0.12 |
| Gatekeeper | 0.84 | 0.16 |
| Speckled Wood | 0.84 | 0.16 |
| Small White | 0.83 | 0.17 |
| Brimstone | 0.71 | 0.29 |
| Meadow Brown | 0.66 | 0.34 |
| Comma | 0.52 | 0.48 |
| Ringlet | 0.42 | 0.58 |
| Common Blue | 0.41 | 0.59 |
| Painted Lady | 0.24 | 0.76 |

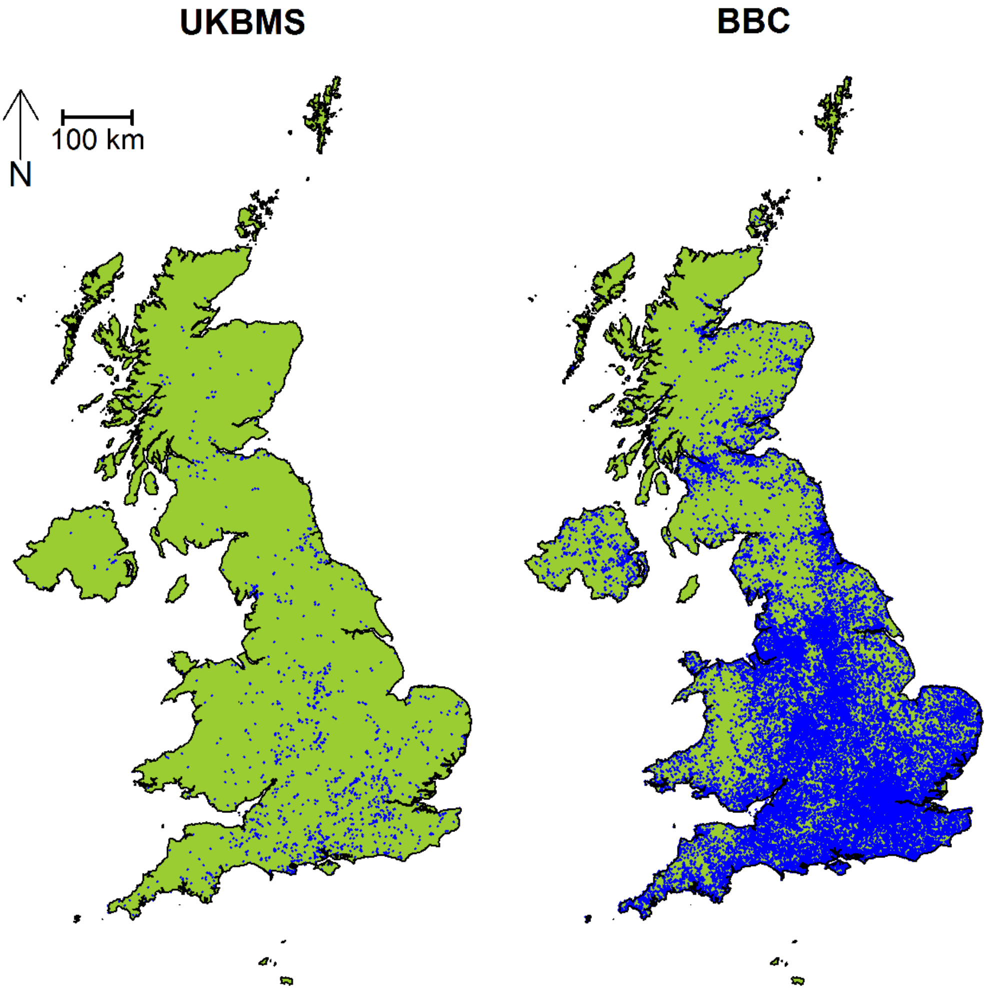


**Fig. A4.1** Locations of UK Butterfly Monitoring Scheme (UKBMS= 1,462 transects) and Big Butterfly Count (BBC= 65,197 1km squares surveyed) counts from 2011-2014 (shown in blue).

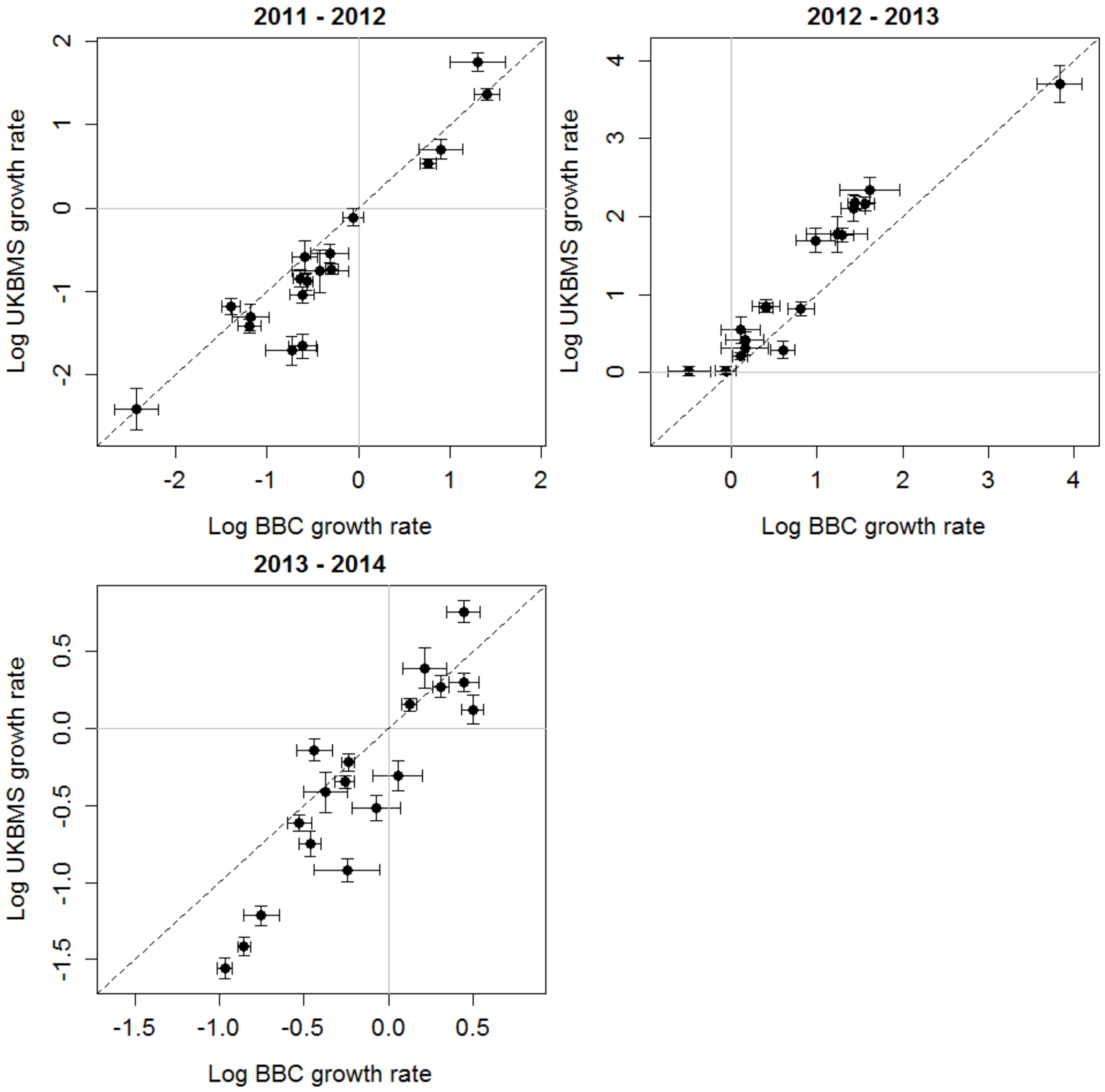
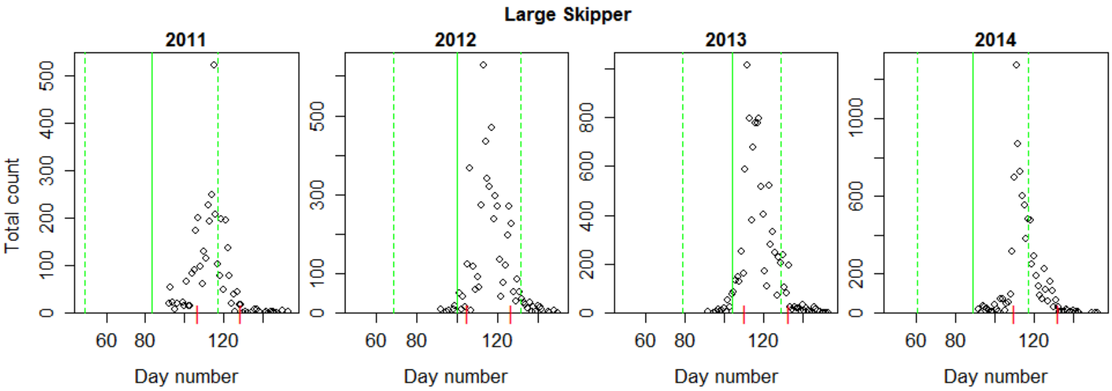
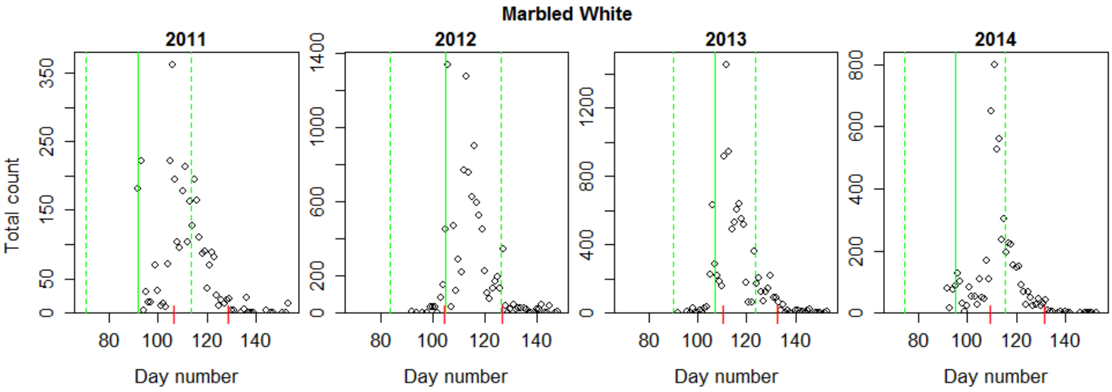
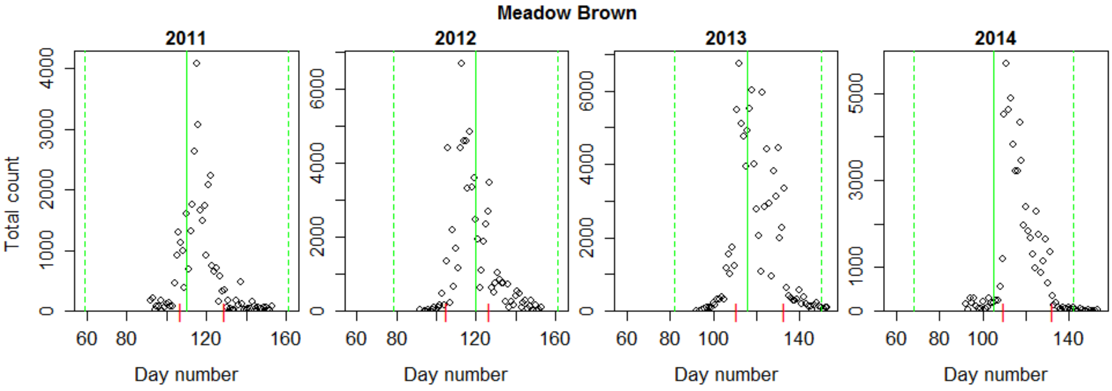
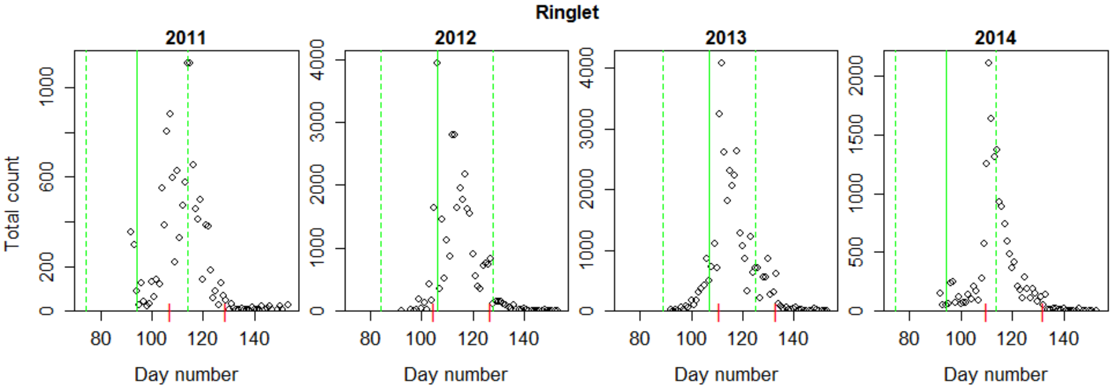


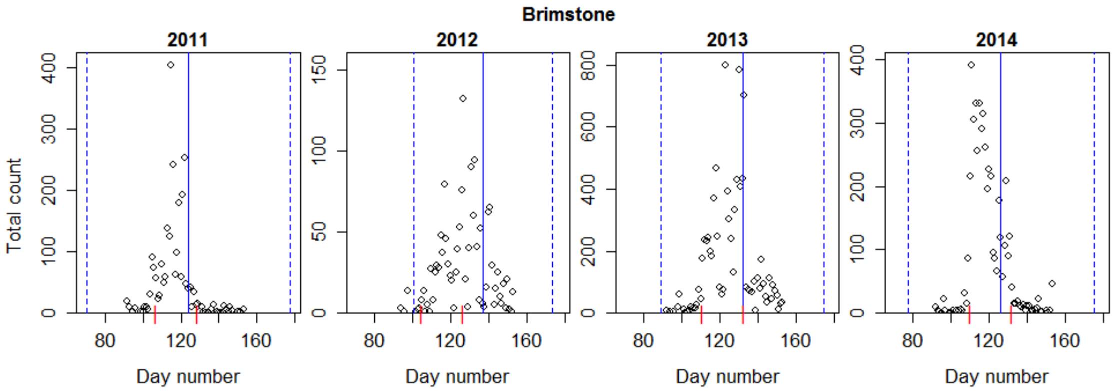
**Fig. A4.2** Comparison of estimated year-to-year log growth rates from the Big Butterfly Count (BBC) and UK Butterfly Monitoring Scheme (UKBMS) for 2011 to 2014. Error bars represent 95% confidence intervals. Solid grey lines represent zero growth and dashed lines represents equal growth rates from the BBC and UKBMS.


Large Skipper
2011
2012
2013
2014
Total count
0
100
200
300
400
500
0
100
300
500
0
200
400
600
800
0
200
600
1000
60
80
120
Day number

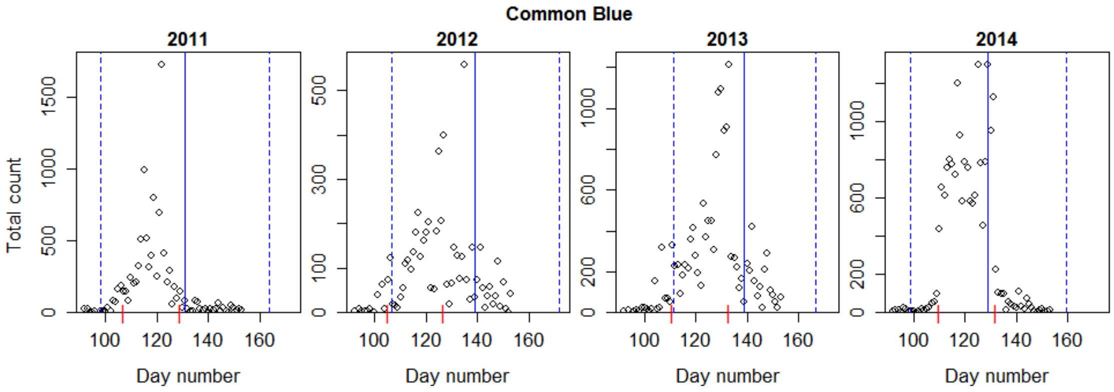

Marbled White
2011
2012
2013
2014
Total count
0
50
150
250
350
0
200
600
1000
1400
0
400
800
1200
0
200
400
600
800
80
100
120
140
Day number


Meadow Brown
2011
2012
2013
2014
Total count
0
1000
2000
3000
4000
0
2000
4000
6000
0
1000
3000
5000
60
80
100
120
140
160
Day number

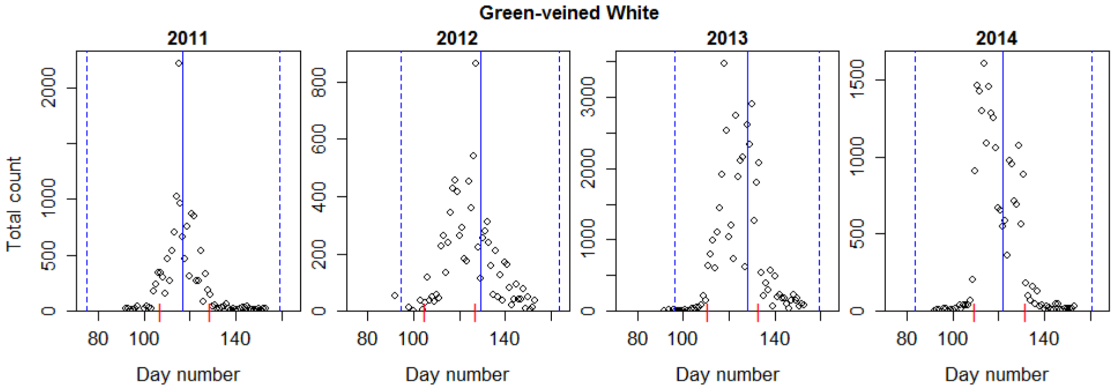

Ringlet
2011
2012
2013
2014
Total count
0
200
600
1000
0
1000
2000
3000
4000
0
500
1000
1500
2000
80
100
120
140
Day number



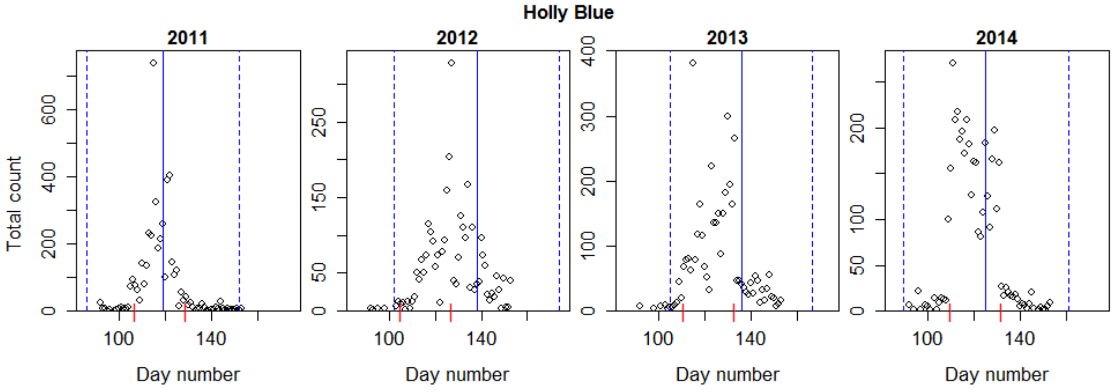
Brimstone



Common Blue



Green-veined White



Holly Blue

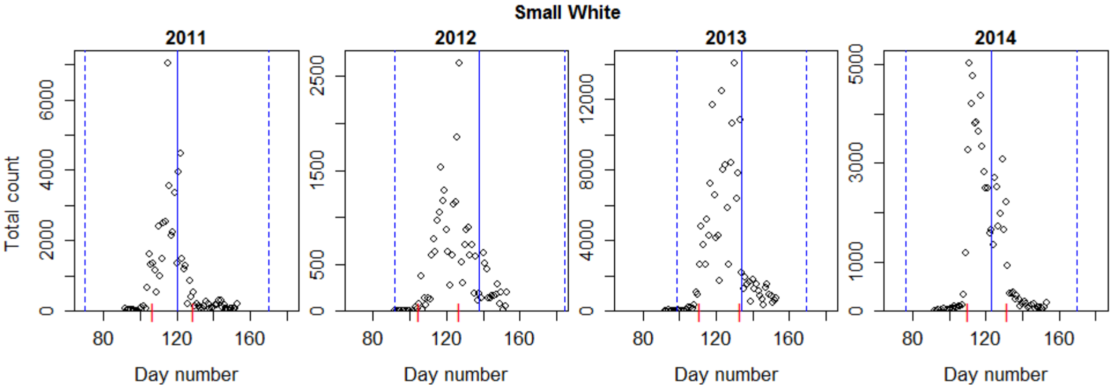


**Fig. A4.3** Total counts of nine species from Big Butterfly Count data per day in each year, where day 1 is 1st April (vertical lines, mean flight dates estimated from a GAI; dashed lines, twice SD; green = 1st brood ; blue = 2nd brood; red lines on x-axis show the official BBC survey period for each year).

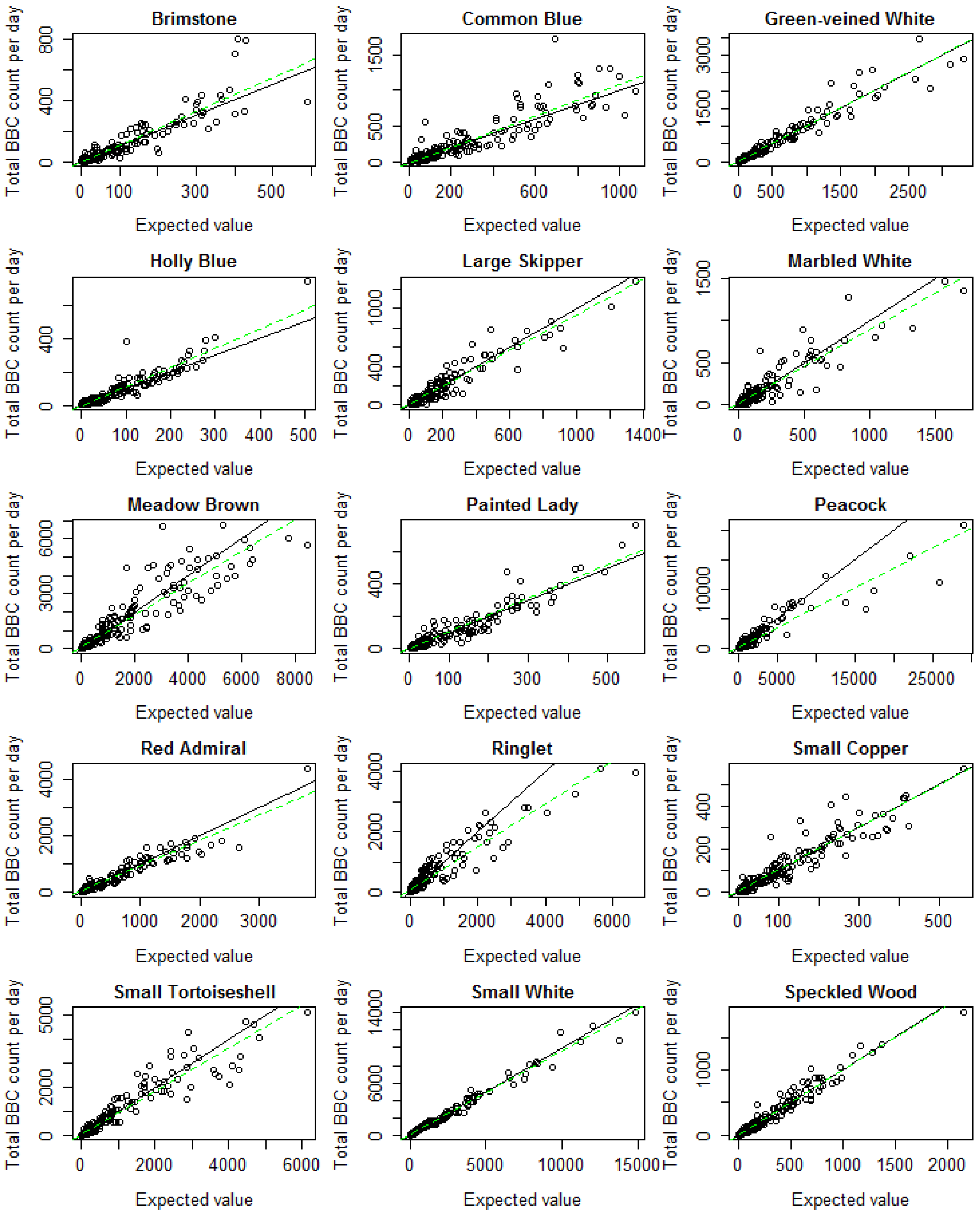


**Fig. A4.4** Total counts for each butterfly species from Big Butterfly Count (BBC) data per day versus the expected value from a negative-binomial GLM in which the response variable is the total count per day and measures of effort (log number of counts made) and phenology (from the corresponding GAI curve) are covariates (black line, equal expected values and total counts; green dashed line, fitted linear regression through the points).

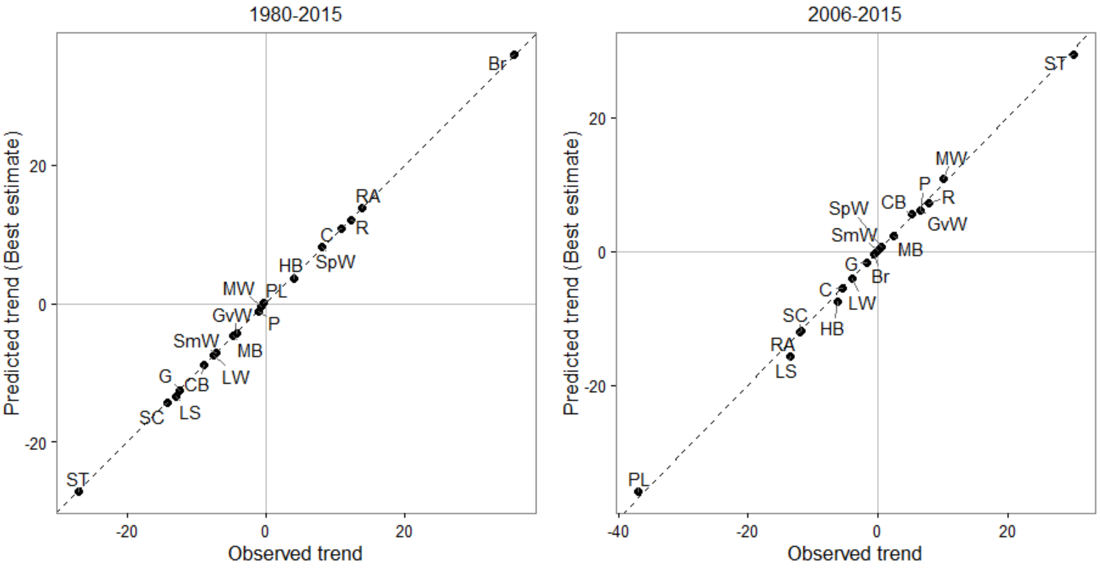


**Fig. A4.5** Comparison of linear trends in relative abundance from the GAI index, where 2015 is from observed data or predicted from the model with the best estimate of the index in 2015 (solid grey lines, 0% change in relative abundance; dashed line, equal population trends). Abbreviations are for species common names (Table A4.2).

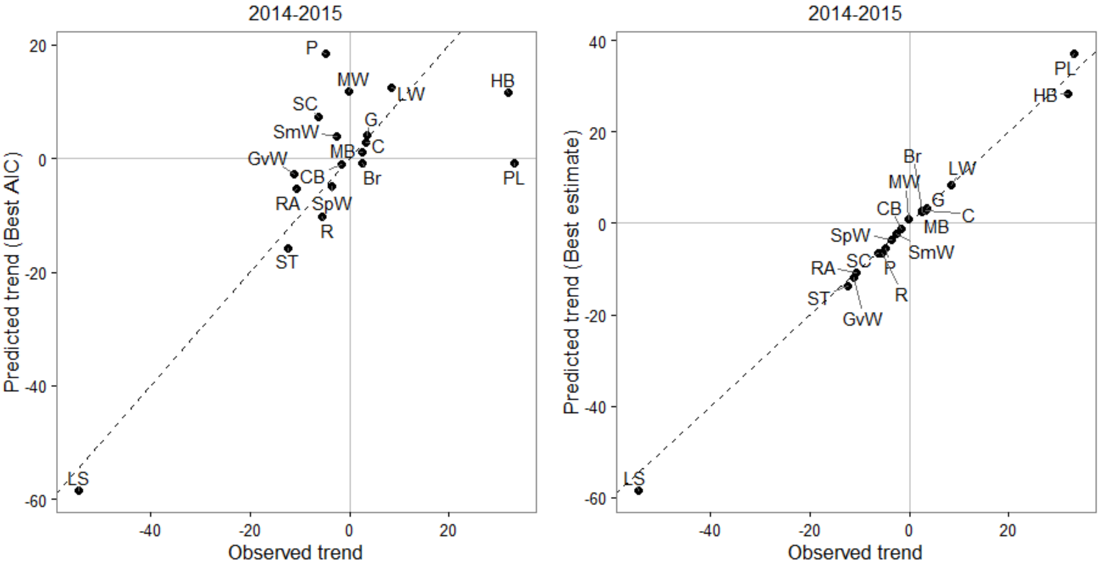


**Fig. A4.6** Comparison of percentage changes in relative abundance 2014-2015, where 2015 is from observed data or predicted from the model with the best estimate of the index in 2015 (solid grey lines, 0% change in relative abundance; dashed line, equal population trends). Abbreviations are for species common names (Table A4.2).

## Appendix 5: Supporting Information for Chapter 5: Insect population trends and the IUCN Red List process

**Table A5.1** UK butterfly species' 10-year population trends over six overlapping date periods, and the median (MT), lower (LQ) and upper quartile (UQ) values of these trends. Shading indicates Red List classification against IUCN Criterion A2 (reduction in population size) thresholds: dark shading = Critically Endangered (population decrease ≥80%), intermediate shading = Endangered (population decrease ≥50%), light shading = Vulnerable (population decrease ≥30%). * common migrants in the UK.

| **Taxon name** | **2001-2010** | **2002-2011** | **2003-2012** | **2004-2013** | **2005-2014** | **2006-2015** | **LQ** | **MT** | **UQ** |
|---|---|---|---|---|---|---|---|---|---|
| *Aglais io* | -9 | -17 | -30 | -6 | 21 | 33 | -15 | -7.5 | 14.25 |
| *Aglais urticae* | -72 | -75 | -77 | -28 | 146 | 216 | -74.25 | -50 | 102.5 |
| *Anthocharis cardamines* | 19 | 36 | 59 | 32 | 59 | 66 | 33 | 47.5 | 59 |
| *Apatura iris* | -15 | -24 | -55 | -19 | -35 | -32 | -34.25 | -28 | -20.25 |
| *Aphantopus hyperantus* | 51 | 38 | 14 | 38 | 72 | 64 | 38 | 44.5 | 60.75 |
| *Argynnis adippe* | -69 | -64 | -81 | -74 | 0 | 46 | -72.75 | -66.5 | -16 |
| *Argynnis aglaja* | 99 | 88 | 12 | 30 | 18 | 6 | 13.5 | 24 | 73.5 |
| *Argynnis paphia* | 137 | 80 | -2 | -3 | 6 | 8 | 0 | 7 | 62 |
| *Aricia agestis* | 53 | 44 | -35 | -34 | -11 | 8 | -28.25 | -1.5 | 35 |
| *Aricia artaxerxes* | -44 | -20 | -27 | -9 | 6 | 19 | -25.25 | -14.5 | 2.25 |
| *Boloria euphrosyne* | -41 | -11 | 10 | 23 | 45 | 87 | -5.75 | 16.5 | 39.5 |
| *Boloria selene* | 16 | 40 | 17 | 6 | 3 | 5 | 5.25 | 11 | 16.75 |
| *Callophrys rubi* | -12 | -15 | -26 | -40 | -34 | -2 | -32 | -20.5 | -12.75 |
| *Celastrina argiolus* | -29 | -28 | -23 | -60 | -61 | -31 | -52.75 | -30 | -28.25 |
| *Coenonympha pamphilus* | 5 | 7 | -22 | 0 | 18 | 27 | 1.25 | 6 | 15.25 |
| *Coenonympha tullia* | 171 | 130 | -6 | -60 | -49 | -56 | -54.25 | -27.5 | 96 |
| *Colias croceus** | -59 | -97 | -98 | -88 | -57 | -32 | -94.75 | -73.5 | -57.5 |
| *Cupido minimus* | 37 | 83 | 0 | -25 | -27 | -19 | -23.5 | -9.5 | 27.75 |
| *Erebia aethiops* | -50 | -44 | -5 | 40 | 24 | 23 | -34.25 | 9 | 23.75 |
| *Erynnis tages* | 3 | 31 | 49 | 57 | 69 | 90 | 35.5 | 53 | 66 |
| *Euphydryas aurinia* | 4 | 42 | -52 | -66 | -64 | -62 | -63.5 | -57 | -10 |
| *Favonius quercus* | -9 | -15 | -8 | -25 | -10 | -17 | -16.5 | -12.5 | -9.25 |
| *Gonepteryx rhamni* | 3 | -5 | -30 | -30 | -1 | 37 | -23.75 | -3 | 2 |
| *Hamearis lucina* | -71 | -51 | -58 | 24 | 67 | 80 | -56.25 | -13.5 | 56.25 |
| *Hesperia comma* | -33 | -19 | -55 | -23 | 12 | 53 | -30.5 | -21 | 4.25 |
| *Hipparchia semele* | 26 | 43 | 18 | 25 | 10 | -7 | 12 | 21.5 | 25.75 |
| *Lasiommata megera* | -57 | -62 | -67 | -56 | -25 | -17 | -60.75 | -56.5 | -32.75 |
| *Leptidea sinapis* | -61 | -55 | -59 | -56 | -18 | 41 | -58.25 | -55.5 | -27.25 |
| *Limenitis camilla* | 80 | 21 | -44 | -50 | -45 | -43 | -44.75 | -43.5 | 5 |
| *Lycaena phlaeas* | 42 | 30 | -16 | -19 | -19 | -46 | -19 | -17.5 | 18.5 |

| | | | | | | | | |
|---|---|---|---|---|---|---|---|---|
| *Maniola jurtina* | -23 | -30 | -31 | -25 | -15 | 13 | -28.75 | -24 | -17 |
| *Melanargia galathea* | -17 | -16 | -28 | -16 | 25 | 62 | -16.75 | -16 | 14.75 |
| *Melitaea athalia* | -72 | -72 | -76 | -82 | -79 | -84 | -81.25 | -77.5 | -73 |
| *Ochlodes sylvanus* | -14 | 3 | -18 | -16 | 23 | 41 | -15.5 | -5.5 | 18 |
| *Pararge aegeria* | 33 | 12 | -20 | -9 | 4 | -5 | -8 | -0.5 | 10 |
| *Phengaris arion* | 23 | -7 | 4 | -11 | -20 | 25 | -10 | -1.5 | 18.25 |
| *Pieris brassicae* | 54 | 8 | -40 | -1 | -28 | -20 | -26 | -10.5 | 5.75 |
| *Pieris napi* | 21 | 20 | -16 | 26 | 72 | 38 | 20.25 | 23.5 | 35 |
| *Pieris rapae* | -5 | -14 | -55 | 10 | 9 | -1 | -11.75 | -3 | 6.5 |
| *Plebejus argus* | -30 | -15 | -40 | 2 | -9 | -6 | -26.25 | -12 | -6.75 |
| *Polygonia c-album* | 11 | -6 | -39 | -25 | -28 | -26 | -27.5 | -25.5 | -10.75 |
| *Polyommatus bellargus* | 99 | -1 | -46 | -23 | -43 | -46 | -45.25 | -33 | -6.5 |
| *Polyommatus coridon* | 21 | 15 | 13 | 77 | 55 | 66 | 16.5 | 38 | 63.25 |
| *Polyommatus icarus* | 16 | -11 | -52 | -27 | 1 | 30 | -23 | -5 | 12.25 |
| *Pyrgus malvae* | -11 | 23 | 63 | 12 | 0 | -5 | -3.75 | 6 | 20.25 |
| *Pyronia tithonus* | -37 | -46 | -62 | -67 | -44 | -6 | -58 | -45 | -38.75 |
| *Satyrium pruni* | 334 | 319 | -40 | -89 | -87 | -78 | -84.75 | -59 | 229.25 |
| *Satyrium w-album* | -26 | -22 | -69 | -74 | -77 | -74 | -74 | -71.5 | -36.75 |
| *Thecla betulae* | -53 | -50 | -75 | -75 | -58 | -5 | -70.75 | -55.5 | -50.75 |
| *Thymelicus acteon* | -93 | -92 | -88 | -53 | 39 | 200 | -91 | -70.5 | 16 |
| *Thymelicus lineola* | -90 | -92 | -91 | -81 | -66 | 16 | -90.75 | -85.5 | -69.75 |
| *Thymelicus sylvestris* | -71 | -66 | -66 | -31 | 27 | 121 | -66 | -48.5 | 12.5 |
| *Vanessa atalanta** | -27 | -13 | -25 | -14 | -40 | -45 | -36.75 | -26 | -16.75 |
| *Vanessa cardui** | -40 | -84 | -94 | -89 | -84 | -88 | -88.75 | -86 | -84 |

**Table A5.2** UK macro-moth 10-year population trends for the most recent 10-year period (t) for each of 431 species, and preceding 10-year periods each starting one year earlier than the previous (t-1, t-2, t-3, t-4) and the Red List categories appropriate to each trend according to IUCN Criterion A2 (reduction in population size). Trend values given in bold indicate statistically significant trends at *p* < 0.05. These classifications do not represent the final outcomes of a full Red List process.

| **Taxon** | **Period t** | | **Period t-1** | | **Period t-2** | | **Period t-3** | | **Period t-4** | |
|---|---|---|---|---|---|---|---|---|---|---|
| *Abraxas grossulariata* | -47.8 | | **-75.0** | EN | **-79.0** | EN | -63.8 | | -20.6 | |
| *Abraxas sylvata* | 23.2 | | 50.8 | | 116.0 | | 41.2 | | 2.8 | |
| *Abrostola tripartita* | -8.4 | | 0.3 | | -21.3 | | -35.9 | | 0.4 | |
| *Abrostola triplasia* | 39.9 | | 7.0 | | -21.2 | | -58.4 | | -61.5 | |
| *Acasis viretata* | -28.0 | | 6.4 | | 22.2 | | 15.7 | | -16.6 | |
| *Achlya flavicornis* | -40.8 | | 10.6 | | 57.6 | | 30.1 | | 13.0 | |
| *Acronicta alni* | -51.3 | | **-75.5** | EN | -78.4 | | -19.6 | | 34.1 | |
| *Acronicta psi* | -39.5 | | -35.6 | | -55.9 | | **-68.6** | EN | **-66.9** | EN |
| *Acronicta rumicis* | **-69.9** | EN | **-58.3** | EN | **-57.9** | EN | -54.7 | | -11.3 | |
| *Aethalura punctulata* | -42.8 | | -25.9 | | -21.8 | | 23.9 | | 118.9 | |
| *Agriopis leucophaearia* | -45.8 | | -4.9 | | 100.8 | | **301.0** | | **337.4** | |
| *Agrochola circellaris* | 23.8 | | 67.0 | | 76.9 | | 6.4 | | 109.4 | |
| *Agrochola helvola* | -18.0 | | -5.6 | | 35.5 | | -23.2 | | -0.6 | |
| *Agrochola litura* | 31.9 | | 25.8 | | 30.6 | | -26.5 | | -3.5 | |
| *Agrochola lota* | 7.0 | | 14.1 | | -3.4 | | -0.4 | | 15.6 | |
| *Agrochola lychnidis* | 181.6 | | 70.3 | | -10.3 | | -38.3 | | -21.1 | |
| *Agrochola macilenta* | 4.3 | | -10.6 | | 12.5 | | 6.5 | | 59.9 | |
| *Agrotis clavis* | 15.4 | | 10.0 | | 1.2 | | 33.6 | | 35.7 | |
| *Agrotis exclamationis* | -15.8 | | -39.8 | | -52.7 | | -21.5 | | 10.9 | |
| *Agrotis puta* | -55.9 | | -48.2 | | -51.5 | | -51.0 | | 1.5 | |
| *Agrotis segetum* | -40.2 | | -62.1 | | -72.4 | | -52.0 | | 105.3 | |
| *Agrotis vestigialis* | 11.3 | | 73.2 | | -20.5 | | 81.0 | | 43.5 | |
| *Alcis jubata* | -52.0 | | 5.1 | | 31.3 | | 38.5 | | 15.6 | |
| *Alcis repandata* | -1.5 | | 3.1 | | 10.7 | | 11.4 | | 27.8 | |
| *Allophyes oxyacanthae* | 8.8 | | 45.0 | | 49.4 | | **91.4** | | **92.6** | |
| *Alsophila aescularia* | 1.3 | | 10.7 | | 0.1 | | 11.8 | | 12.3 | |
| *Amphipoea crinanensis* | 14.3 | | -37.3 | | -49.7 | | -1.4 | | 91.6 | |
| *Amphipoea oculea* | -25.1 | | -9.3 | | -22.9 | | -13.1 | | 36.2 | |
| *Amphipyra berbera* | -26.5 | | 138.4 | | 327.4 | | 460.1 | | **605.4** | |
| *Amphipyra pyramidea* | -63.7 | | **-75.0** | EN | **-74.1** | EN | -56.3 | | -24.7 | |
| *Amphipyra tragopoginis* | -27.4 | | -52.6 | | **-60.2** | EN | **-61.9** | EN | **-46.5** | VU |
| *Anaplectoides prasina* | -3.7 | | -31.0 | | -21.3 | | -36.7 | | -4.7 | |
| *Anarta trifolii* | -46.8 | | -82.4 | | -86.8 | | -27.0 | | -43.2 | |
| *Anorthoa munda* | 5.5 | | 46.7 | | 57.4 | | **110.0** | | 45.0 | |
| *Anticlea derivata* | -18.2 | | -13.8 | | -33.7 | | 1.7 | | 38.7 | |
| *Antitype chi* | 209.6 | | 65.1 | | 56.9 | | 11.5 | | -29.9 | |
| *Apamea anceps* | -67.2 | | -54.0 | | -53.2 | | -35.1 | | 55.6 | |
| *Apamea crenata* | -26.6 | | 6.0 | | -24.4 | | -17.3 | | 25.0 | |
| *Apamea lithoxylaea* | 20.4 | | 50.1 | | 3.8 | | 10.5 | | 57.1 | |

| | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|
| *Apamea monoglypha* | 15.7 | | 2.1 | | -40.8 | | -26.0 | | -0.9 | |
| *Apamea remissa* | 2.4 | | 28.8 | | -14.8 | | -27.9 | | 29.4 | |
| *Apamea scolopacina* | **155.0** | | **112.1** | | **61.6** | | 35.1 | | 15.5 | |
| *Apamea sordens* | **-70.6** | EN | -38.8 | | -22.0 | | 11.1 | | **123.9** | |
| *Apeira syringaria* | -49.4 | | -58.0 | | -59.6 | | **-65.5** | EN | -29.9 | |
| *Aplocera efformata* | -36.4 | | -37.3 | | -44.0 | | -22.0 | | 38.7 | |
| *Aplocera plagiata* | -69.9 | | -51.7 | | -20.2 | | -31.8 | | -37.7 | |
| *Apocheima hispidaria* | -1.7 | | 181.8 | | **414.3** | | 198.9 | | 52.9 | |
| *Aporophyla lueneburgensis* | 160.8 | | 125.7 | | 23.4 | | **-73.8** | EN | -62.1 | |
| *Aporophyla lutulenta* | -6.4 | | -18.7 | | -38.3 | | -25.9 | | 58.4 | |
| *Aporophyla nigra* | -40.3 | | -43.7 | | **-47.1** | VU | **-47.8** | VU | -12.6 | |
| *Apterogenum ypsillon* | -85.3 | | -59.4 | | 52.2 | | 92.6 | | **560.0** | |
| *Arctia caja* | 60.8 | | 44.6 | | -7.2 | | -19.9 | | -42.0 | |
| *Aspitates ochrearia* | **-90.9** | CR | **-96.7** | CR | **-87.1** | CR | **-83.0** | CR | **-67.0** | EN |
| *Asteroscopus sphinx* | -3.0 | | 10.9 | | **87.2** | | **113.7** | | **126.2** | |
| *Asthena albulata* | -40.8 | | -57.7 | | **-70.2** | EN | -56.0 | | -5.3 | |
| *Atethmia centrago* | -25.3 | | -18.1 | | 33.5 | | 5.3 | | 54.5 | |
| *Autographa bractea* | -8.9 | | 33.4 | | 30.0 | | -15.6 | | -45.9 | |
| *Autographa jota* | 155.8 | | **245.1** | | **175.7** | | 49.7 | | 5.6 | |
| *Autographa pulchrina* | 84.3 | | 121.4 | | 121.3 | | 26.0 | | -14.0 | |
| *Axylia putris* | 10.4 | | -19.1 | | -40.2 | | -40.1 | | -24.9 | |
| *Bena bicolorana* | -73.6 | | **-88.1** | CR | -1.6 | | 85.3 | | 28.6 | |
| *Biston betularia* | **-78.2** | EN | **-58.6** | EN | **-56.6** | EN | **-55.4** | EN | **-45.4** | VU |
| *Biston strataria* | 19.9 | | 58.1 | | 46.2 | | 53.0 | | **69.5** | |
| *Brachylomia viminalis* | 59.4 | | 17.6 | | 13.6 | | 28.8 | | 29.0 | |
| *Bryophila domestica* | -4.4 | | -24.0 | | -43.0 | | **-58.4** | EN | **-64.2** | EN |
| *Bupalus piniaria* | 166.1 | | -4.6 | | -42.7 | | -31.0 | | -24.1 | |
| *Cabera exanthemata* | 3.5 | | 33.9 | | 39.0 | | 5.5 | | 48.2 | |
| *Cabera pusaria* | 20.8 | | 16.4 | | -19.4 | | **-39.4** | VU | -30.4 | |
| *Calliteara pudibunda* | -14.4 | | -23.9 | | -43.5 | | -24.9 | | 31.0 | |
| *Campaea margaritata* | 27.0 | | **57.8** | | 43.7 | | 21.9 | | 33.4 | |
| *Camptogramma bilineata* | **-37.8** | VU | -37.4 | | -32.9 | | -26.8 | | 6.4 | |
| *Caradrina clavipalpis* | **-85.5** | CR | **-92.7** | CR | **-93.3** | CR | **-84.5** | CR | **-75.8** | EN |
| *Caradrina morpheus* | **70.7** | | 36.9 | | 0.6 | | -6.1 | | -7.0 | |
| *Catarhoe rubidata* | **-91.9** | CR | **-85.8** | CR | **-60.9** | EN | -36.6 | | 118.9 | |
| *Catocala nupta* | 41.7 | | 60.1 | | 122.2 | | 54.6 | | 146.3 | |
| *Celaena haworthii* | 198.2 | | **224.7** | | **275.7** | | 27.9 | | 21.7 | |
| *Ceramica pisi* | -49.4 | | -58.0 | | **-70.4** | EN | -57.2 | | -41.6 | |
| *Cerapteryx graminis* | -41.0 | | 28.5 | | 83.8 | | 31.5 | | 75.6 | |
| *Cerastis leucographa* | 69.5 | | **256.4** | | **357.0** | | 118.7 | | -0.5 | |
| *Cerastis rubricosa* | -17.5 | | -12.0 | | -42.7 | | 4.1 | | 9.2 | |
| *Charanyca trigrammica* | 4.0 | | 4.4 | | 2.6 | | 116.6 | | 96.1 | |
| *Chesias legatella* | -40.9 | | -51.6 | | -34.7 | | -59.8 | | -17.1 | |
| *Chesias rufata* | 58.7 | | 15.9 | | 69.2 | | -6.5 | | -73.1 | |
| *Chiasmia clathrata* | -40.9 | | -43.0 | | **-64.8** | EN | **-68.0** | EN | -51.4 | |
| *Chloroclysta miata* | -2.1 | | 52.5 | | 70.8 | | 21.9 | | 40.1 | |
| *Chloroclysta siterata* | -6.3 | | 29.6 | | 54.1 | | 37.8 | | 63.2 | |
| *Chloroclystis v-ata* | 4.0 | | 37.1 | | 1.5 | | 18.1 | | 104.3 | |
| *Cidaria fulvata* | -11.6 | | -6.7 | | -18.4 | | -51.3 | | -30.1 | |
| *Cilix glaucata* | -21.8 | | -21.0 | | **-44.1** | VU | **-52.1** | EN | -32.1 | |
| *Cirrhia icteritia* | -27.2 | | -2.7 | | 24.2 | | 27.2 | | 63.1 | |

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|---|---|---|---|---|---|---|---|---|---|
| *Cleorodes lichenaria* | 6.7 | | 0.3 | | -26.8 | | -28.4 | | -11.9 |
| *Clostera curtula* | -32.2 | | 6.7 | | 22.6 | | **79.3** | | **73.5** |
| *Coenobia rufa* | -21.1 | | -35.5 | | **-73.5** | EN | **-84.4** | CR | -68.0 |
| *Coenotephria salicata* | -19.9 | | 24.2 | | 9.0 | | -31.3 | | 14.8 |
| *Colocasia coryli* | -21.5 | | 5.3 | | -30.1 | | -20.3 | | 6.1 |
| *Colostygia pectinataria* | -7.2 | | 4.2 | | 40.9 | | **48.2** | | 41.5 |
| *Colotois pennaria* | -1.3 | | 23.3 | | 11.7 | | 28.6 | | 23.7 |
| *Comibaena bajularia* | **-82.1** | CR | -74.2 | | -71.6 | | -43.2 | | 214.3 |
| *Conistra ligula* | 36.8 | | 44.9 | | 27.3 | | 78.6 | | 124.3 |
| *Conistra rubiginea* | -43.7 | | 73.6 | | 17.1 | | -3.6 | | 37.3 |
| *Conistra vaccinii* | 56.2 | | 46.9 | | 44.7 | | 60.0 | | 61.4 |
| *Cosmia affinis* | 14.4 | | 76.1 | | 58.1 | | 51.1 | | 97.9 |
| *Cosmia pyralina* | -40.5 | | -11.6 | | -14.5 | | 7.0 | | **154.3** |
| *Cosmia trapezina* | -40.4 | | -2.0 | | 50.8 | | 22.2 | | 77.1 |
| *Cosmorhoe ocellata* | 6.8 | | -6.4 | | -35.9 | | **-51.0** | EN | -29.1 |
| *Craniophora ligustri* | 26.7 | | 11.8 | | -19.6 | | -46.9 | | -30.5 |
| *Crocallis elinguaria* | -21.4 | | -6.3 | | -7.9 | | 2.8 | | 19.7 |
| *Cybosia mesomella* | 45.6 | | 24.1 | | -3.0 | | 2.4 | | -0.7 |
| *Cyclophora albipunctata* | -34.7 | | -45.5 | | **-70.0** | EN | -39.6 | | -22.4 |
| *Cyclophora linearia* | -74.2 | | -27.0 | | -53.1 | | -58.9 | | -20.5 |
| *Cyclophora punctaria* | **-60.2** | EN | -44.0 | | **-62.8** | EN | **-66.8** | EN | -40.1 |
| *Cymatophorina diluta* | **-98.5** | CR | **-96.9** | CR | **-91.5** | CR | -47.7 | | -10.0 |
| *Dasypolia templi* | 178.5 | | **413.0** | | **476.6** | | 82.5 | | -28.8 |
| *Deilephila elpenor* | 3.6 | | -37.5 | | **-70.7** | EN | **-48.5** | VU | -39.8 |
| *Deileptenia ribeata* | -24.9 | | -12.9 | | -29.0 | | **-54.6** | EN | -23.4 |
| *Deltote pygarga* | **680.5** | | 255.0 | | 78.8 | | 32.2 | | 79.5 |
| *Deltote uncula* | 9.7 | | 1.5 | | 25.8 | | -21.8 | | 23.4 |
| *Denticucullus pygmina* | 16.0 | | 66.0 | | **93.7** | | 44.9 | | 66.8 |
| *Diachrysia chrysitis* | 10.5 | | 20.3 | | 40.3 | | 13.5 | | 6.1 |
| *Diacrisia sannio* | 26.4 | | 5.4 | | -39.3 | | -35.0 | | -49.9 |
| *Diaphora mendica* | -37.2 | | -32.1 | | -13.1 | | 89.9 | | **126.8** |
| *Diarsia brunnea* | 32.5 | | 20.6 | | 5.5 | | 11.4 | | 28.2 |
| *Diarsia dahlii* | 68.8 | | 42.8 | | 38.0 | | -0.9 | | 23.9 |
| *Diarsia mendica* | 61.3 | | 18.4 | | -6.8 | | -10.4 | | -14.7 |
| *Diarsia rubi* | 13.1 | | -4.6 | | -28.0 | | -21.5 | | 12.2 |
| *Diloba caeruleocephala* | -72.0 | | -26.8 | | -31.7 | | 13.6 | | -21.5 |
| *Drepana falcataria* | -20.0 | | -24.8 | | **-48.9** | VU | **-49.0** | VU | -13.0 |
| *Drymonia dodonaea* | 5.9 | | -2.9 | | -20.8 | | -18.0 | | -3.4 |
| *Drymonia ruficornis* | -74.7 | | -76.1 | | -78.5 | | -23.3 | | 24.2 |
| *Dryobotodes eremita* | -37.4 | | -24.4 | | -22.1 | | -30.6 | | -5.1 |
| *Dypterygia scabriuscula* | -13.2 | | 1.3 | | **-74.6** | EN | **-85.6** | CR | -64.3 |
| *Dyscia fagaria* | 13.4 | | -41.6 | | 20.4 | | -17.9 | | -35.6 |
| *Dysstroma citrata* | 4.0 | | 34.1 | | 37.0 | | -14.5 | | -14.1 |
| *Dysstroma truncata* | 39.4 | | 73.7 | | 84.6 | | 19.2 | | 27.8 |
| *Earophila badiata* | 2.7 | | 8.4 | | 8.3 | | 63.7 | | 51.4 |
| *Ecliptopera silaceata* | 15.5 | | 85.6 | | 60.6 | | 9.6 | | 28.7 |
| *Ectropis* sp. | 6.8 | | 8.3 | | 5.8 | | 17.8 | | 38.4 |
| *Eilema complana* | -6.5 | | -42.7 | | -43.9 | | -47.0 | | -25.4 |
| *Eilema depressa* | 96.9 | | 73.1 | | 29.4 | | 5.4 | | 37.4 |
| *Eilema griseola* | 61.2 | | 80.6 | | 33.3 | | 12.6 | | 19.8 |
| *Eilema lurideola* | -3.2 | | -4.2 | | -19.4 | | 2.6 | | 12.7 |
| *Electrophaes corylata* | 32.2 | | 49.5 | | 11.0 | | 4.5 | | 24.5 |

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|---|---|---|---|---|---|---|---|---|---|---|
| *Ennomos alniaria* | **-50.9** | EN | **-49.4** | VU | -35.6 | | -43.0 | | -22.0 | |
| *Ennomos erosaria* | 111.9 | | 40.4 | | -30.1 | | -45.7 | | -22.4 | |
| *Ennomos fuscantaria* | 2.0 | | -22.9 | | -43.9 | | **-72.2** | EN | **-59.2** | EN |
| *Ennomos quercinaria* | 36.4 | | **81.9** | | 77.5 | | -3.1 | | -28.0 | |
| *Entephria caesiata* | 333.0 | | **534.4** | | 351.8 | | 141.1 | | -39.9 | |
| *Epione repandaria* | 13.1 | | 32.5 | | 25.0 | | 23.8 | | 44.7 | |
| *Epirrhoe alternata* | 17.3 | | 5.7 | | -22.6 | | **-50.7** | EN | -39.4 | |
| *Epirrhoe galiata* | -50.4 | | -31.5 | | -61.3 | | **-84.8** | CR | **-76.3** | EN |
| *Epirrhoe rivata* | **-47.5** | VU | -30.8 | | 3.4 | | -7.1 | | **31.0** | |
| *Epirrita autumnata* | 1.8 | | 7.5 | | -19.8 | | -13.4 | | -23.4 | |
| *Epirrita christyi* | 9.2 | | 33.6 | | 54.4 | | 31.7 | | 30.4 | |
| *Epirrita dilutata* | -16.5 | | -7.5 | | 12.7 | | 16.8 | | 23.8 | |
| *Epirrita filigrammaria* | -42.2 | | -54.1 | | -54.6 | | **-78.5** | EN | -57.1 | |
| *Eremobia ochroleuca* | **175.5** | | **101.8** | | 33.5 | | -4.2 | | -12.2 | |
| *Euchoeca nebulata* | -68.6 | | **-75.1** | EN | **-72.9** | EN | **-83.3** | CR | -33.3 | |
| *Eugnorisma depuncta* | **-78.0** | EN | **-77.1** | EN | -55.5 | | -65.2 | | -39.1 | |
| *Eugnorisma glareosa* | -13.6 | | -54.8 | | -50.4 | | -27.9 | | -12.9 | |
| *Eulithis populata* | 100.9 | | 97.0 | | 46.8 | | -9.9 | | -3.5 | |
| *Eulithis prunata* | -15.9 | | 40.7 | | 40.1 | | -31.8 | | -45.1 | |
| *Eulithis testata* | 8.0 | | -11.4 | | -30.2 | | **-55.2** | EN | -36.4 | |
| *Euphyia biangulata* | **-63.2** | EN | **-55.7** | EN | -53.1 | | -30.3 | | 21.2 | |
| *Euphyia unangulata* | **-59.7** | EN | **-61.4** | EN | -57.2 | | -51.7 | | -35.9 | |
| *Eupithecia abbreviata* | -42.2 | | 3.0 | | 28.7 | | 69.1 | | **90.3** | |
| *Eupithecia absinthiata* | 35.2 | | 13.4 | | -23.6 | | -48.6 | | 29.3 | |
| *Eupithecia assimilata* | -52.2 | | -25.5 | | -25.8 | | -26.9 | | 38.5 | |
| *Eupithecia centaureata* | -50.7 | | -36.2 | | -34.1 | | **-58.5** | EN | -37.1 | |
| *Eupithecia distinctaria* | **-78.9** | EN | -70.5 | | **-78.5** | EN | -64.9 | | **-82.3** | CR |
| *Eupithecia exiguata* | **-53.5** | EN | -45.2 | | -40.4 | | -25.1 | | 50.2 | |
| *Eupithecia haworthiata* | -3.3 | | -20.0 | | -51.5 | | -59.9 | | 4.7 | |
| *Eupithecia icterata* | 24.4 | | 12.5 | | -4.1 | | -33.6 | | -31.4 | |
| *Eupithecia innotata* | **-81.3** | CR | **-84.7** | CR | -68.3 | | -25.3 | | 57.1 | |
| *Eupithecia intricata* | -4.1 | | -32.1 | | -22.2 | | -29.0 | | -26.4 | |
| *Eupithecia inturbata* | 30.0 | | -32.8 | | -30.0 | | -26.1 | | -6.5 | |
| *Eupithecia lariciata* | **-71.2** | EN | -42.1 | | 43.4 | | **309.7** | | **504.5** | |
| *Eupithecia linariata* | **-88.2** | CR | **-89.7** | CR | **-88.7** | CR | **-88.7** | CR | -75.4 | |
| *Eupithecia nanata* | -43.1 | | -24.3 | | -32.6 | | -43.4 | | -45.7 | |
| *Eupithecia plumbeolata* | 319.9 | | 336.1 | | 75.5 | | 35.3 | | 131.5 | |
| *Eupithecia pulchellata* | 1.6 | | 37.8 | | -5.5 | | -11.8 | | 51.7 | |
| *Eupithecia pusillata* | -36.0 | | -24.3 | | 2.5 | | -16.7 | | 56.0 | |
| *Eupithecia simpliciata* | 65.5 | | -57.1 | | **-82.4** | CR | -75.5 | | -55.6 | |
| *Eupithecia subfuscata* | -46.0 | | -39.1 | | -47.7 | | -21.0 | | 157.8 | |
| *Eupithecia subumbrata* | 20.1 | | 80.5 | | 49.8 | | 1.4 | | 25.1 | |
| *Eupithecia succenturiata* | -6.4 | | -4.8 | | -22.2 | | **-62.2** | EN | -58.0 | |
| *Eupithecia tantillaria* | -40.5 | | -1.0 | | -1.0 | | 71.4 | | 10.2 | |
| *Eupithecia tenuiata* | -26.7 | | -13.4 | | 6.0 | | 20.8 | | 202.6 | |
| *Eupithecia tripunctaria* | -38.7 | | -6.5 | | -12.7 | | -15.6 | | 119.6 | |
| *Eupithecia trisignaria* | **334.9** | | **463.8** | | **373.6** | | -23.7 | | -28.1 | |
| *Eupithecia valerianata* | -50.1 | | **-80.3** | CR | **-83.3** | CR | -65.8 | | -3.5 | |
| *Eupithecia venosata* | **-86.0** | CR | **-88.1** | CR | -76.2 | | -78.6 | | -68.9 | |
| *Eupithecia virgaureata* | -59.3 | | -64.5 | | -62.6 | | -58.5 | | 157.6 | |
| *Eupithecia vulgata* | -22.5 | | 0.5 | | -11.0 | | 21.9 | | **72.7** | |
| *Euplexia lucipara* | -0.4 | | 4.1 | | -17.3 | | -38.5 | | -22.6 | |

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|---|---|---|---|---|---|---|---|---|---|---|
| *Euproctis chrysorrhoea* | 11.2 | | -18.5 | | -30.5 | | -32.9 | | 55.8 | |
| *Euproctis similis* | -30.7 | | -32.2 | | -45.6 | | **-49.0** | VU | -25.9 | |
| *Eupsilia transversa* | -10.6 | | -3.4 | | 13.2 | | 73.3 | | **140.9** | |
| *Euthrix potatoria* | 15.5 | | -1.4 | | -32.5 | | **-40.5** | VU | -29.7 | |
| *Furcula furcula* | -64.2 | | **-79.6** | EN | -65.4 | | -78.1 | | -49.0 | |
| *Gandaritis pyraliata* | 35.1 | | 32.8 | | 27.6 | | -7.0 | | -4.7 | |
| *Geometra papilionaria* | -7.5 | | -6.3 | | -19.0 | | -25.6 | | -10.3 | |
| *Gnophos obfuscata* | -92.4 | | -92.9 | | -87.0 | | -72.5 | | -63.5 | |
| *Gortyna flavago* | -20.4 | | 11.5 | | 42.9 | | 7.8 | | 49.6 | |
| *Graphiphora augur* | **182.3** | | 92.3 | | 87.7 | | 3.8 | | 19.9 | |
| *Griposia aprilina* | -6.6 | | -6.1 | | 13.9 | | -15.0 | | 6.4 | |
| *Gymnoscelis rufifasciata* | **-68.4** | EN | **-67.6** | EN | **-74.3** | EN | **-67.4** | EN | -3.9 | |
| *Habrosyne pyritoides* | 50.0 | | 71.4 | | 96.9 | | 1.4 | | 19.1 | |
| *Hada plebeja* | -52.4 | | -27.2 | | -38.6 | | -18.6 | | -25.9 | |
| *Hadena confusa* | -46.4 | | **-91.9** | CR | **-97.3** | CR | **-94.8** | CR | **-95.5** | CR |
| *Hadena perplexa* | -4.0 | | 56.7 | | -42.4 | | -31.6 | | -62.0 | |
| *Hecatera bicolorata* | -39.3 | | 2.5 | | -26.2 | | -41.6 | | 1.3 | |
| *Helotropha leucostigma* | 5.6 | | -20.6 | | -21.2 | | -45.5 | | 10.2 | |
| *Hemistola chrysoprasaria* | **-62.4** | EN | **-49.4** | VU | **-47.8** | VU | -13.0 | | 8.8 | |
| *Hemithea aestivaria* | 2.1 | | -12.1 | | -0.7 | | -9.3 | | 44.3 | |
| *Hepialus humuli* | -6.6 | | -31.5 | | -21.6 | | -23.5 | | 3.1 | |
| *Herminia grisealis* | -3.9 | | -8.3 | | -16.2 | | -24.9 | | 2.2 | |
| *Herminia tarsipennalis* | 34.3 | | 50.7 | | 88.8 | | 53.4 | | 69.6 | |
| *Hoplodrina alsines* | 29.0 | | 24.6 | | 15.0 | | 45.0 | | 71.4 | |
| *Hoplodrina ambigua* | -29.8 | | -39.9 | | -57.2 | | -47.4 | | 42.1 | |
| *Hoplodrina blanda* | 81.8 | | 40.9 | | 0.2 | | 2.6 | | -14.2 | |
| *Horisme tersata* | -29.0 | | -33.4 | | -23.8 | | -29.3 | | 1.3 | |
| *Horisme vitalbata* | -49.7 | | -57.4 | | **-62.9** | EN | **-63.5** | EN | -13.6 | |
| *Hydraecia micacea* | 34.8 | | 41.1 | | 27.2 | | -2.5 | | 46.6 | |
| *Hydrelia flammeolaria* | 81.2 | | -56.0 | | **-78.4** | EN | **-80.9** | CR | -67.6 | |
| *Hydrelia sylvata* | 24.8 | | -3.1 | | 66.2 | | 28.0 | | **202.0** | |
| *Hydria undulata* | 18.9 | | -15.4 | | -30.4 | | -38.2 | | 17.9 | |
| *Hydriomena furcata* | 134.2 | | 153.9 | | 87.9 | | 14.6 | | 7.8 | |
| *Hydriomena impluviata* | -23.5 | | -34.4 | | -59.4 | | -56.7 | | -1.5 | |
| *Hydriomena ruberata* | 204.4 | | 189.9 | | 70.8 | | -52.1 | | -52.0 | |
| *Hylaea fasciaria* | 21.9 | | **44.3** | | 25.3 | | 31.4 | | 19.4 | |
| *Hypena crassalis* | -68.7 | | -69.1 | | -29.1 | | 27.0 | | 111.5 | |
| *Hypena proboscidalis* | -4.7 | | 15.6 | | 26.0 | | -15.6 | | 6.9 | |
| *Hypomecis punctinalis* | **-49.4** | VU | **-47.7** | VU | **-63.4** | EN | **-61.1** | EN | -42.7 | |
| *Hypomecis roboraria* | -12.4 | | -5.1 | | -10.2 | | 18.0 | | 9.9 | |
| *Hyppa rectilinea* | 54.4 | | 60.2 | | 28.1 | | 31.5 | | 0.5 | |
| *Idaea aversata* | -12.2 | | -3.5 | | -19.4 | | 0.8 | | 17.0 | |
| *Idaea biselata* | 7.1 | | 21.4 | | 2.0 | | -6.5 | | 2.0 | |
| *Idaea dimidiata* | -30.1 | | -19.4 | | -24.4 | | -21.4 | | -5.9 | |
| *Idaea emarginata* | 1.4 | | 27.3 | | -5.4 | | 14.8 | | 34.4 | |
| *Idaea fuscovenosa* | -37.0 | | -39.4 | | -33.0 | | -1.1 | | 43.5 | |
| *Idaea rusticata* | 88.1 | | **126.8** | | **137.7** | | **164.8** | | **193.4** | |
| *Idaea seriata* | -36.3 | | -18.5 | | -30.5 | | -38.3 | | -6.4 | |
| *Idaea subsericeata* | -57.4 | | **-72.3** | EN | -70.9 | | **-77.5** | EN | -42.5 | |
| *Idaea trigeminata* | -32.2 | | -27.8 | | -39.6 | | -8.8 | | 71.3 | |
| *Jodis lactearia* | 57.4 | | 72.1 | | 42.4 | | 13.8 | | 41.5 | |

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|---|---|---|---|---|---|---|---|---|---|---|
| *Korscheltellus fusconebulosa* | 19.8 | | -6.6 | | 71.9 | | 60.6 | | 90.9 | |
| *Korscheltellus lupulina* | -29.0 | | -31.4 | | -34.1 | | -10.9 | | -17.7 | |
| *Lacanobia contigua* | **-71.7** | EN | -57.9 | | -27.3 | | 48.1 | | 146.4 | |
| *Lacanobia oleracea* | -48.0 | | **-66.6** | EN | **-77.0** | EN | **-77.1** | EN | -50.3 | |
| *Lacanobia suasa* | -14.9 | | -29.8 | | -2.7 | | -25.2 | | 110.5 | |
| *Lacanobia thalassina* | -1.2 | | -1.7 | | -27.5 | | -24.2 | | 15.9 | |
| *Lampropteryx otregiata* | 19.4 | | 57.9 | | 59.9 | | 23.5 | | 45.8 | |
| *Lampropteryx suffumata* | 76.7 | | 126.7 | | 68.6 | | 70.8 | | 82.4 | |
| *Laothoe populi* | -11.5 | | -24.8 | | -25.0 | | -14.4 | | -8.3 | |
| *Larentia clavaria* | **-46.8** | VU | **-49.2** | VU | -30.7 | | -30.3 | | -19.5 | |
| *Lasiocampa quercus* | 13.5 | | -60.2 | | -66.9 | | -81.2 | | -69.2 | |
| *Laspeyria flexula* | 44.9 | | 97.4 | | 109.5 | | 161.1 | | **540.4** | |
| *Lateroligia ophiogramma* | -50.4 | | -67.6 | | -75.0 | | -74.0 | | -41.0 | |
| *Leucania comma* | **-69.7** | EN | **-72.3** | EN | **-59.2** | EN | -49.2 | | -18.7 | |
| *Ligdia adustata* | 55.6 | | 43.7 | | -9.5 | | -29.6 | | 49.0 | |
| *Lithophane leautieri* | 17.5 | | 18.3 | | 51.8 | | -4.0 | | -22.4 | |
| *Lithophane ornitopus* | -29.6 | | -30.5 | | -17.1 | | 21.2 | | 205.0 | |
| *Lithophane socia* | -76.6 | | -80.5 | | -62.2 | | -64.9 | | -31.9 | |
| *Litoligia literosa* | -5.8 | | -8.0 | | -14.5 | | -13.2 | | **-63.1** | EN |
| *Lobophora halterata* | -19.4 | | -26.8 | | -42.1 | | -29.9 | | -20.3 | |
| *Lomaspilis marginata* | 3.0 | | -5.2 | | -14.1 | | -8.7 | | 22.8 | |
| *Lomographa bimaculata* | -55.2 | | -36.9 | | -47.3 | | -42.2 | | 24.4 | |
| *Lomographa temerata* | 58.8 | | 42.2 | | 14.7 | | -4.6 | | 47.5 | |
| *Luperina testacea* | -0.6 | | -2.8 | | -7.3 | | 0.1 | | 19.9 | |
| *Lycia hirtaria* | **102.9** | | **95.4** | | 67.4 | | **138.6** | | **141.8** | |
| *Lycophotia porphyrea* | -3.2 | | -20.1 | | -39.3 | | -32.8 | | -34.7 | |
| *Lygephila pastinum* | 9.7 | | 16.7 | | 17.4 | | 79.0 | | 232.3 | |
| *Lymantria monacha* | 60.9 | | 74.2 | | 41.7 | | 15.9 | | 27.3 | |
| *Macaria alternata* | -46.8 | | -51.4 | | **-62.0** | EN | -52.7 | | 27.7 | |
| *Macaria liturata* | **-73.4** | EN | **-77.1** | EN | **-77.3** | EN | **-72.5** | EN | -56.6 | |
| *Macaria notata* | 52.7 | | -0.7 | | -49.0 | | -65.6 | | -41.8 | |
| *Macrothylacia rubi* | -15.1 | | -2.5 | | -22.8 | | -10.5 | | 59.2 | |
| *Malacosoma neustria* | -66.0 | | -46.9 | | -41.1 | | -16.9 | | 44.9 | |
| *Mamestra brassicae* | -44.1 | | -49.4 | | -54.1 | | **-74.1** | EN | -60.0 | |
| *Melanchra persicariae* | 31.0 | | -0.8 | | -45.4 | | **-76.7** | EN | -68.9 | |
| *Melanthia procellata* | 41.3 | | 99.2 | | **133.8** | | 54.2 | | 35.6 | |
| *Menophra abruptaria* | -40.8 | | 8.2 | | 5.4 | | 27.7 | | **85.8** | |
| *Mesapamea secalis/didyma* | -1.7 | | -30.9 | | -42.6 | | -48.2 | | -18.9 | |
| *Mesoleuca albicillata* | -37.3 | | -47.4 | | -46.9 | | **-65.7** | EN | -52.9 | |
| *Mesoligia furuncula* | -51.0 | | -19.3 | | 2.7 | | -7.0 | | 21.0 | |
| *Mesotyp didymata* | 133.7 | | 49.0 | | 10.1 | | **-67.2** | EN | **-74.5** | EN |
| *Miltochrista miniata* | 27.3 | | -1.6 | | -3.3 | | -27.1 | | 1.4 | |
| *Mimas tiliae* | -77.3 | | **-86.0** | CR | **-90.2** | CR | -82.6 | | -70.2 | |
| *Mniotype adusta* | 17.4 | | 52.8 | | 34.2 | | 72.9 | | 127.6 | |
| *Mythimna albipuncta* | **215.6** | | **198.9** | | 76.3 | | 78.7 | | **191.5** | |
| *Mythimna conigera* | 12.8 | | 30.0 | | 33.5 | | **75.7** | | **103.7** | |
| *Mythimna ferrago* | **133.2** | | **140.5** | | **93.4** | | 52.8 | | 34.5 | |
| *Mythimna impura* | -4.8 | | -20.6 | | **-35.8** | VU | **-32.9** | VU | -19.3 | |
| *Mythimna pallens* | **-76.4** | EN | **-83.8** | CR | **-86.2** | CR | **-92.0** | CR | **-75.2** | EN |
| *Naenia typica* | 113.9 | | **345.8** | | 212.8 | | 109.2 | | 40.4 | |
| *Noctua comes* | -10.8 | | -29.7 | | -24.6 | | -14.7 | | 29.5 | |

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|---|---|---|---|---|---|---|---|---|---|---|
| *Noctua fimbriata* | -12.1 | | -15.8 | | -19.5 | | -4.2 | | 40.3 | |
| *Noctua interjecta* | -80.0 | | -69.5 | | **-82.2** | CR | **-91.8** | CR | **-82.5** | CR |
| *Noctua pronuba* | -33.9 | | **-44.8** | VU | **-47.0** | VU | **-49.5** | VU | **-34.1** | VU |
| *Nola confusalis* | -35.0 | | -31.1 | | -40.4 | | -11.9 | | **74.6** | |
| *Nola cucullatella* | **-61.7** | EN | -31.4 | | -9.3 | | -26.2 | | 7.7 | |
| *Notodonta dromedarius* | **-61.3** | EN | **-50.7** | EN | **-50.0** | EN | **-56.4** | EN | -35.5 | |
| *Notodonta ziczac* | -4.6 | | 0.7 | | -23.1 | | **-45.1** | VU | -29.0 | |
| *Nudaria mundana* | 55.3 | | **144.1** | | **155.4** | | 80.7 | | 67.3 | |
| *Nycteola revayana* | **-65.7** | EN | **-65.4** | EN | **-67.6** | EN | **-64.9** | EN | 5.9 | |
| *Nyctobrya muralis* | -26.7 | | -50.0 | | -15.5 | | -67.5 | | -76.7 | |
| *Ochropacha duplaris* | -48.2 | | -28.8 | | -36.8 | | -43.5 | | -7.6 | |
| *Ochropleura plecta* | -25.3 | | -15.9 | | -21.4 | | -32.3 | | -2.1 | |
| *Odezia atrata* | 87.9 | | 63.3 | | 57.4 | | -46.0 | | -63.3 | |
| *Odontopera bidentata* | -35.8 | | 13.8 | | 16.1 | | 49.5 | | 32.0 | |
| *Odontosia carmelita* | 28.6 | | 23.5 | | 23.8 | | 5.2 | | 12.1 | |
| *Oligia fasciuncula* | -20.1 | | -14.2 | | 14.9 | | 12.4 | | 85.8 | |
| *Oligia latruncula* | 11.3 | | 34.0 | | 7.6 | | 2.0 | | 18.4 | |
| *Oligia strigilis* | **143.0** | | 85.5 | | 49.3 | | 25.7 | | 37.5 | |
| *Oligia versicolor* | 58.1 | | 34.4 | | -3.5 | | -14.0 | | 8.8 | |
| *Omphaloscelis lunosa* | -2.6 | | -46.3 | | **-71.7** | EN | **-76.2** | EN | -55.6 | |
| *Opisthograptis luteolata* | 9.9 | | 22.2 | | 15.5 | | 4.7 | | 22.4 | |
| *Orthonama vittata* | 31.0 | | -38.6 | | -48.7 | | -53.6 | | -10.6 | |
| *Orthosia cerasi* | 44.8 | | 62.5 | | 16.3 | | 44.0 | | 5.5 | |
| *Orthosia cruda* | -4.1 | | 61.0 | | 132.7 | | **235.1** | | **93.8** | |
| *Orthosia gothica* | 34.5 | | 64.9 | | 25.0 | | 42.0 | | 9.3 | |
| *Orthosia gracilis* | 75.5 | | 78.6 | | 73.2 | | 101.1 | | 52.2 | |
| *Orthosia incerta* | -19.9 | | 14.2 | | -7.6 | | 27.2 | | 24.8 | |
| *Orthosia populeti* | -10.8 | | 52.3 | | 274.2 | | 305.7 | | 252.5 | |
| *Ourapteryx sambucaria* | 24.6 | | 22.2 | | 33.9 | | -17.4 | | 22.3 | |
| *Pachycnemia hippocastanaria* | 103.1 | | **248.1** | | **201.1** | | **138.3** | | **520.0** | |
| *Panolis flammea* | -65.5 | | -69.2 | | **-85.9** | CR | -50.9 | | -47.0 | |
| *Papestra biren* | 14.9 | | 97.3 | | 46.4 | | -23.6 | | -51.3 | |
| *Paradarisa consonaria* | -35.5 | | -36.9 | | -9.5 | | 214.9 | | 154.8 | |
| *Parascotia fuliginaria* | -32.6 | | -27.3 | | **-47.4** | VU | **-46.1** | VU | -37.2 | |
| *Parectropis similaria* | 1.0 | | -40.7 | | **-75.4** | EN | **-69.2** | EN | **-63.3** | EN |
| *Pasiphila debiliata* | 236.9 | | 194.7 | | 36.1 | | -9.1 | | 134.9 | |
| *Pasiphila rectangulata* | -54.9 | | -57.5 | | -50.0 | | -50.5 | | -16.5 | |
| *Pennithera firmata* | -31.4 | | -19.0 | | 28.4 | | 57.2 | | **134.5** | |
| *Peribatodes rhomboidaria* | **-44.5** | VU | -27.2 | | -11.0 | | 5.2 | | 27.9 | |
| *Peridea anceps* | -2.6 | | -12.2 | | **-53.0** | EN | **-53.7** | EN | **-47.8** | VU |
| *Perizoma affinitata* | 33.3 | | 59.1 | | 23.7 | | -14.5 | | 6.0 | |
| *Perizoma albulata* | -35.1 | | -1.5 | | -27.6 | | -54.6 | | -19.4 | |
| *Perizoma alchemillata* | 55.5 | | 71.4 | | 70.7 | | 60.2 | | 49.6 | |
| *Perizoma bifaciata* | **-92.5** | CR | **-88.4** | CR | **-75.9** | EN | -12.9 | | 36.3 | |
| *Perizoma blandiata* | **-76.5** | EN | **-73.4** | EN | **-82.3** | CR | **-85.1** | CR | -55.4 | |
| *Perizoma flavofasciata* | 12.4 | | 18.8 | | -9.1 | | -30.1 | | -5.7 | |
| *Petrophora chlorosata* | -19.1 | | -17.7 | | -22.5 | | -10.1 | | 42.4 | |
| *Phalera bucephala* | 21.5 | | -8.8 | | -31.5 | | -37.4 | | -32.0 | |
| *Pheosia gnoma* | -8.9 | | -12.3 | | -13.9 | | 34.4 | | **69.8** | |
| *Pheosia tremula* | -43.5 | | 10.3 | | 14.7 | | 50.0 | | 7.5 | |
| *Phileremé transversata* | 85.9 | | 70.5 | | 86.2 | | 42.5 | | **186.6** | |

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|---|---|---|---|---|---|---|---|---|---|---|
| *Phileremе vetulata* | 34.0 | | 33.3 | | -31.4 | | -33.7 | | -47.9 | |
| *Phlogophora meticulosa* | -45.3 | | -46.8 | | -46.5 | | -48.4 | | **-54.8** | EN |
| *Photedes fluxa* | 8.1 | | 12.1 | | -21.6 | | 71.0 | | 79.6 | |
| *Photedes minima* | -4.8 | | -15.7 | | **-25.0** | | -18.9 | | -20.4 | |
| *Phragmatobia fuliginosa* | -40.6 | | -17.3 | | -19.2 | | -18.4 | | 65.1 | |
| *Phymatopus hecta* | 228.1 | | 122.8 | | 42.2 | | 29.0 | | -6.1 | |
| *Phytometra viridaria* | 165.6 | | 50.9 | | -50.6 | | **-73.1** | EN | -69.7 | |
| *Plagodis dolabraria* | -7.6 | | 30.0 | | -25.8 | | -27.8 | | 13.0 | |
| *Plagodis pulveraria* | 36.4 | | 95.7 | | 70.5 | | 52.3 | | 89.1 | |
| *Plemyria rubiginata* | -34.6 | | -27.3 | | -25.2 | | -56.1 | | -54.5 | |
| *Plusia festucae* | -15.2 | | 20.7 | | -0.5 | | -40.6 | | -36.8 | |
| *Plusia putnami* | **-90.8** | CR | **-80.2** | CR | **-69.1** | EN | -43.2 | | 0.1 | |
| *Poecilocampa populi* | 18.8 | | 22.7 | | 17.2 | | 35.0 | | 34.4 | |
| *Polia nebulosa* | **169.9** | | 106.2 | | 46.8 | | 18.3 | | -20.5 | |
| *Polymixis flavicincta* | **-86.9** | CR | **-91.1** | CR | **-90.8** | CR | **-90.1** | CR | **-86.8** | CR |
| *Polymixis lichenea* | **-93.8** | CR | **-89.0** | CR | **-49.0** | VU | -15.5 | | 12.9 | |
| *Polyploca ridens* | -53.6 | | 18.4 | | 78.4 | | **155.3** | | **177.7** | |
| *Pseudoips prasinana* | -33.3 | | -9.7 | | -4.1 | | -9.7 | | -21.9 | |
| *Pseudoterpna pruinata* | 0.8 | | -2.7 | | -47.8 | | -55.2 | | -23.9 | |
| *Pterapherapteryx sexalata* | -49.2 | | -47.5 | | -62.7 | | -57.5 | | 7.8 | |
| *Pterostoma palpina* | -37.1 | | -24.2 | | -29.0 | | -31.9 | | 9.1 | |
| *Ptilodon capucina* | 23.9 | | 12.8 | | 14.0 | | -18.3 | | 2.7 | |
| *Ptilodon cucullina* | -40.5 | | -18.6 | | 12.9 | | 22.8 | | 34.5 | |
| *Rivula sericealis* | -57.4 | | -50.0 | | -46.3 | | -47.9 | | 8.4 | |
| *Rusina ferruginea* | -20.4 | | -19.5 | | -19.6 | | -5.8 | | 8.8 | |
| *Saturnia pavonia* | **-84.2** | CR | **-83.3** | CR | -30.9 | | 11.0 | | **284.3** | |
| *Schrankia taenialis* | -53.8 | | 51.7 | | -13.9 | | -56.2 | | -14.3 | |
| *Scopula floslactata* | **-58.4** | EN | -35.9 | | **-46.4** | VU | **-41.4** | VU | -3.6 | |
| *Scopula imitaria* | **-59.9** | EN | **-60.5** | EN | **-62.9** | EN | **-63.8** | EN | -36.1 | |
| *Scopula immutata* | -17.7 | | 6.9 | | 3.1 | | -3.6 | | -9.5 | |
| *Scopula marginepunctata* | **-70.3** | EN | **-64.6** | EN | -50.6 | | -65.0 | | -58.5 | |
| *Scopula ternata* | **-80.5** | CR | **-72.5** | EN | **-71.2** | EN | -23.8 | | 11.7 | |
| *Scotopteryx chenopodiata* | 47.5 | | 66.2 | | 42.6 | | -0.5 | | -8.3 | |
| *Scotopteryx luridata* | -76.3 | | **-79.3** | EN | -56.0 | | -58.3 | | 6.1 | |
| *Scotopteryx mucronata* | **274.0** | | **752.1** | | **342.1** | | 148.8 | | -33.5 | |
| *Selenia dentaria* | 13.3 | | 66.0 | | 42.9 | | 16.7 | | 19.5 | |
| *Selenia lunularia* | 49.1 | | 71.7 | | -16.1 | | -33.7 | | -14.8 | |
| *Selenia tetralunaria* | 50.5 | | 102.7 | | 89.1 | | 66.1 | | 83.2 | |
| *Sideridis rivularis* | 1.3 | | -12.5 | | -37.0 | | **-64.3** | EN | -52.7 | |
| *Sideridis turbida* | 92.9 | | -0.3 | | 6.2 | | 20.9 | | 45.7 | |
| *Spilosoma lubricipeda* | 6.7 | | -8.0 | | -32.0 | | -23.5 | | -15.6 | |
| *Spilosoma lutea* | 1.7 | | -13.6 | | -29.6 | | -28.8 | | -9.0 | |
| *Stauropus fagi* | **-66.8** | EN | **-69.7** | EN | **-65.5** | EN | **-53.1** | EN | -36.9 | |
| *Stilbia anomala* | -19.3 | | **-52.7** | EN | **-60.4** | EN | -38.5 | | -46.1 | |
| *Subacronicta megacephala* | -82.5 | | -14.3 | | -45.1 | | -20.6 | | 36.5 | |
| *Syngrapha interrogationis* | -66.9 | | -59.0 | | -65.6 | | 15.0 | | -36.5 | |
| *Tethea ocularis* | -38.3 | | -43.4 | | -54.2 | | -32.0 | | 28.2 | |
| *Tethea or* | -72.0 | | -50.3 | | -32.4 | | -27.1 | | -18.9 | |
| *Tetheella fluctuosa* | 64.0 | | 7.4 | | -46.2 | | -39.9 | | -40.8 | |
| *Thalpophila matura* | 3.2 | | 23.1 | | 34.0 | | 24.2 | | **99.8** | |

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|---|---|---|---|---|---|---|---|---|---|---|
| *Thera britannica* | -56.1 | | -25.1 | | 17.8 | | 9.8 | | 173.0 | |
| *Thera juniperata* | -53.0 | | 38.7 | | 98.7 | | 153.7 | | 103.5 | |
| *Thera obeliscata* | **-55.4** | EN | -51.4 | | -45.8 | | -45.1 | | -10.3 | |
| *Theria primaria* | -31.4 | | 6.3 | | 9.8 | | 27.2 | | 19.6 | |
| *Tholera cespitis* | -18.2 | | -26.2 | | 8.6 | | -7.6 | | 35.1 | |
| *Tholera decimalis* | 20.1 | | 96.4 | | **207.7** | | 100.6 | | **209.0** | |
| *Thumatha senex* | 110.8 | | 37.3 | | 25.0 | | 1.5 | | -27.3 | |
| *Thyatira batis* | 3.0 | | 15.4 | | 11.4 | | -24.0 | | 7.2 | |
| *Tiliacea aurago* | -0.1 | | 20.6 | | -2.1 | | 28.9 | | **141.8** | |
| *Timandra comae* | 3.5 | | -2.4 | | -29.9 | | -53.4 | | -5.9 | |
| *Trichiura crataegi* | -50.8 | | -43.5 | | -29.3 | | -16.0 | | 22.5 | |
| *Trichopteryx carpinata* | -27.2 | | -14.2 | | -2.7 | | 30.5 | | **105.9** | |
| *Triodia sylvina* | 23.6 | | 0.7 | | -29.1 | | -40.4 | | -37.2 | |
| *Tyria jacobaeae* | 9.4 | | 3.9 | | -31.9 | | 72.2 | | 259.5 | |
| *Watsonalla binaria* | -56.4 | | -35.3 | | -56.2 | | -54.4 | | 39.1 | |
| *Xanthia togata* | 7.7 | | 1.8 | | 31.2 | | 66.8 | | **86.1** | |
| *Xanthorhoe decoloraria* | **140.8** | | **165.1** | | 45.9 | | -38.2 | | -67.0 | |
| *Xanthorhoe designata* | -10.5 | | 29.2 | | 8.8 | | -28.0 | | -0.3 | |
| *Xanthorhoe ferrugata* | -26.4 | | -5.7 | | -4.7 | | -13.4 | | -10.5 | |
| *Xanthorhoe fluctuata* | -29.0 | | -16.3 | | -26.9 | | **-46.9** | VU | **-38.6** | VU |
| *Xanthorhoe montanata* | 40.4 | | **63.8** | | 54.5 | | 13.9 | | 17.5 | |
| *Xanthorhoe quadrifasciata* | 140.5 | | 143.3 | | **179.7** | | 92.5 | | 67.4 | |
| *Xanthorhoe spadicearia* | 25.0 | | 81.0 | | 80.1 | | 60.9 | | **101.0** | |
| *Xestia agathina* | **-77.6** | EN | -65.5 | | -51.9 | | -47.1 | | 0.0 | |
| *Xestia baja* | 16.5 | | 12.5 | | 28.4 | | 16.3 | | **30.3** | |
| *Xestia castanea* | 42.9 | | 50.7 | | 20.2 | | 21.7 | | 3.1 | |
| *Xestia c-nigrum* | -12.1 | | -17.9 | | -15.9 | | -38.9 | | 0.5 | |
| *Xestia ditrapezium* | 29.7 | | **126.9** | | 80.3 | | 3.6 | | -6.9 | |
| *Xestia sexstrigata* | 75.5 | | -20.4 | | -49.2 | | **-63.1** | EN | -37.7 | |
| *Xestia triangulum* | 48.7 | | 57.1 | | 45.6 | | 31.5 | | 66.4 | |
| *Xestia xanthographa* | -11.2 | | -26.4 | | -20.2 | | -29.2 | | 17.8 | |
| *Xylena solidaginis* | 128.6 | | **539.2** | | 319.1 | | 112.2 | | 193.7 | |
| *Xylena vetusta* | 26.7 | | 54.1 | | 12.8 | | 50.3 | | 58.3 | |
| *Xylocampa areola* | -34.6 | | -21.0 | | -24.2 | | 35.1 | | 38.7 | |

# Appendix 6: Supporting Information for Chapter 6: Opinions of citizen scientists on open access to UK butterfly and moth occurrence data

## Questionnaire to BNM and NMRS regional co-ordinators (referred to as County Recorders)

## Open Access and butterfly and moth recording scheme data

Please complete this short questionnaire as fully as possible and return it by 31st May 2017. *Your specific responses will be anonymous in any results that we share publicly or with partner organisations.*

Name: …………………………… County/VC: ……………………………………

**1.** Do your responses on this questionnaire relate to *(put X in one box)*

Butterfly records only (BNM scheme) ☐

Moth records only (NMRS) ☐

Both ☐

**2.** Given that there is always some access to records (e.g. via published distribution maps), on a scale from 1 (serious reservations) to 10 (strongly in favour), how much are you in favour of open access to butterfly/moth records? *(put X in box below one number)*

| 1 | 2 | 3 | 4 | 5 | 6 | 7 | 8 | 9 | 10 |
|---|---|---|---|---|---|---|---|---|---|
| ☐ | ☐ | ☐ | ☐ | ☐ | ☐ | ☐ | ☐ | ☐ | ☐ |

**3.** Given the balance between maximising use and minimising risk, what spatial resolution would be best for open access butterfly/moth records? *(put X in one box in each column)*

| | Common/widespread species | Threatened species |
|---|---|---|
| Full capture resolution *(the resolution of the raw record)* | ☐ | ☐ |
| 1km square / monad *(1km x 1km OS map square)* | ☐ | ☐ |
| 2km square / tetrad *(2km x 2km OS map square)* | ☐ | ☐ |
| 10km square / hectad *(10km x 10km OS map square)* | ☐ | ☐ |

**4.** Should there be a time lag between records reaching the database and becoming open access? If so, how long should it be? *(put X in one box)*

20 years ☐ 10 years ☐ 5 years ☐ No time lag ☐

**5.** Are there specific species or colonies of species in your area that would require a different, more restrictive approach to that detailed in Q3 and Q4? *(put X in one box)*

Yes ☐ No ☐

If yes, please indicate which/how many species:

**6.** Should BNM and NMRS open access data be provided publicly under a: *(put X in one box)*

Creative Commons Zero licence ☐
*(no acknowledgment and no limit on use)*

Creative Commons with Attribution licence ☐
*(recording scheme must be acknowledged but no limit on data use)*

Creative Commons, with Attribution, Non-commercial licence ☐
*(recording scheme must be acknowledged and data cannot be used for commercial purposes)*

**7.** Comments *(please add anything else you want to tell us re open access and butterfly/moth recording scheme data)*

………………………………………………………………………………………………

………………………………………………………………………………………………

………………………………………………………………………………………………

*Continue overleaf if necessary*

## Covering letter sent with questionnaire to BNM and NMRS regional co-ordinators (referred to as County Recorders)

Dear County Recorder,

*Open Access and butterfly and moth recording scheme data*

This letter is being sent to all County Moth Recorders, County Butterfly Recorders, Butterfly Conservation Branch Chairs and other organisations (e.g. certain Local Environmental Records Centres) directly involved in supplying verified county datasets to the Butterflies for the New Millennium (BNM) recording scheme and the National Moth Recording Scheme (NMRS). The aim is to consult you and gather your views, aspirations and concerns about "open access".

In the last decade, there have been increasing calls amongst the scientific and conservation community for data, including biological records, to be made freely and openly accessible to all. This is part of a much wider shift towards open access of data amongst Government and Society, both in the UK and globally. Many arguments are put forward to support open access, ranging from ethical to economic, and government departments, the statutory conservation agencies and some major wildlife charities such as the British Trust for Ornithology have made their biological records open access (although others have not). Conservation decisions, policy, research and public education are increasingly dependent on open access to biological records. It is likely that many of the recorders who provide the raw records expect and assume that their sightings are widely available and being used to the fullest extent to support conservation. In parallel with this consultation, we will also be seeking general opinions from "grass roots" recorders about open access.

Open access means that records are freely available to all i.e. there is no need to seek permission or to pay for access to the data. However, it does not mean that records have to be available at 'full capture resolution' (i.e. the spatial resolution of the original, raw record) or that users can do anything they like with the data.

To date, Butterfly Conservation has responded very cautiously to the increasing calls for open access to butterfly and moth records. Currently, only historical (pre-2000) NMRS macro-moth records for the UK and all BNM butterfly records for Scotland are open access under non-commercial use licenses (e.g. via the NBN) and neither are available at full capture resolution.

However, as time passes, it is appropriate to review this position and take stock of the current views of key volunteers, partner organisations and recorders, upon whom our butterfly and moth recording schemes depend. The desire to maximise the use of records for conservation and other public benefits (through being more open with access) must be balanced against the desire to protect sites and species from harm and data from commercial exploitation (through being more restrictive with access). In addition, such considerations will vary from species to species and at different spatial resolutions. For example, some rare and threatened species may be at a greater risk of harm than common and widespread species, and records at summary spatial resolution (e.g. 2km x 2km squares) are of far less use to the commercial sector than fine-scale (full capture resolution) records. As compilers of records, we would ask you to be mindful of the expectations of the recorders who submit their sightings to you, but as the organiser of the UK schemes, Butterfly Conservation will take specific note of your opinion.

The aim of this consultation is to help plan the future for access to Butterfly Conservation’s recording scheme datasets (BNM and NMRS) to which you contribute so we need to know your views. I can assure you that Butterfly Conservation’s principal concern is to maintain the fantastic and vital recording schemes in which you play an essential part.

I would be extremely grateful if you could take a few minutes to complete the enclosed questionnaire please (either electronically or on paper) and return it by 31st May. Only those people named as County Recorders on the BNM and NMRS lists and BC Branch Chairs will receive this letter, so if you work as part of a team with other key individuals, please feel free to consult with them and submit responses that reflect the group’s views.

Many thanks and best wishes

Richard Fox
Head of Recording

## Questionnaire to UK butterfly and moth recorders contributing to the BNM and NMRS projects

### Open public access to butterfly and moth records

You’ve been asked to complete this brief survey (which should take less than 5 minutes) because you regularly contribute butterfly and/or moth records via your County Recorder to Butterfly Conservation’s UK recording schemes (Butterflies for the New Millennium and the National Moth Recording Scheme).

Your answers are anonymous and will help Butterfly Conservation to plan future access to the recording scheme datasets. County Recorders have already been consulted, but as a provider of the sightings on which the schemes rely, your opinions are also very important to us.

The records you provide are already used in many ways by Butterfly Conservation and partner organisations (e.g. statutory conservation agencies, local environmental record centres) including assessing species trends, targeting habitat management, informing planning decisions and in collaborative scientific research. However, the records are not currently openly available to the public, policy makers or scientists to view and use without restriction.

We would be very grateful if you would complete the following short survey by 9th June 2017.

**1)** Do you contribute butterfly records, moth records or both to your County Recorders? *

○ Butterfly records

○ Moth records

○ Both butterfly and moth records

**2)** Where is most of your UK butterfly/moth recording done? *

| | |
|---|---|
| ○ England | ○ Scotland |
| ○ Northern Ireland | ○ Wales |

**3)** What is your preference for the public accessibility of your butterfly and/or moth records via the national schemes? *

○ All my records should be publicly accessible in full detail (i.e. the grid reference at which you originally submitted the record) to maximise their use

○ My records of widespread species should be publicly accessible in full detail, but my records of scarce/threatened species to be publicly accessible only at a summary level (i.e. my original detailed grid reference is blurred to a coarser resolution such as tetrad 2km x 2km level) so that precise locations cannot be easily identified

○ All my records should be publicly accessible at summary level (i.e. with the grid reference blurred) so that precise locations cannot be easily identified

**4)** If the UK datasets of butterfly/moth records were openly accessible in full detail to the public, would you be likely to: *

○ Withhold your records from County Recorders and the UK recording schemes?

○ Reduce the precision of the records that you currently provide (e.g. by blurring the grid references of your records before you submit them)?

○ Continue to provide records in exactly the same way?

○ Provide more records to County Recorders and the UK recording schemes?

If you have any other comments that you would like to make on this subject, please use the box below: ☐

**Many thanks for your help!**

**Richard Fox, Head of Recording, Butterfly Conservation**

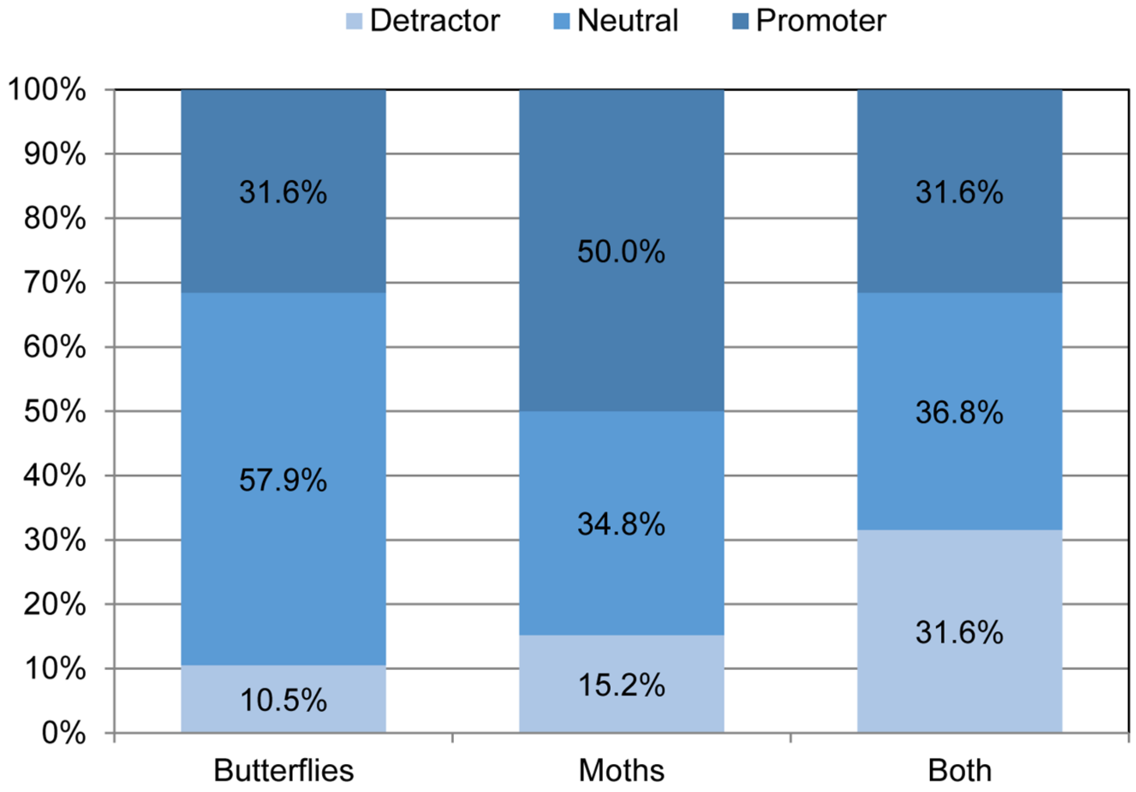

**Fig. A6.1** Levels of general support for open access, assessed by modified NPS categories, among regional co-ordinators for butterflies, moths or both taxa.

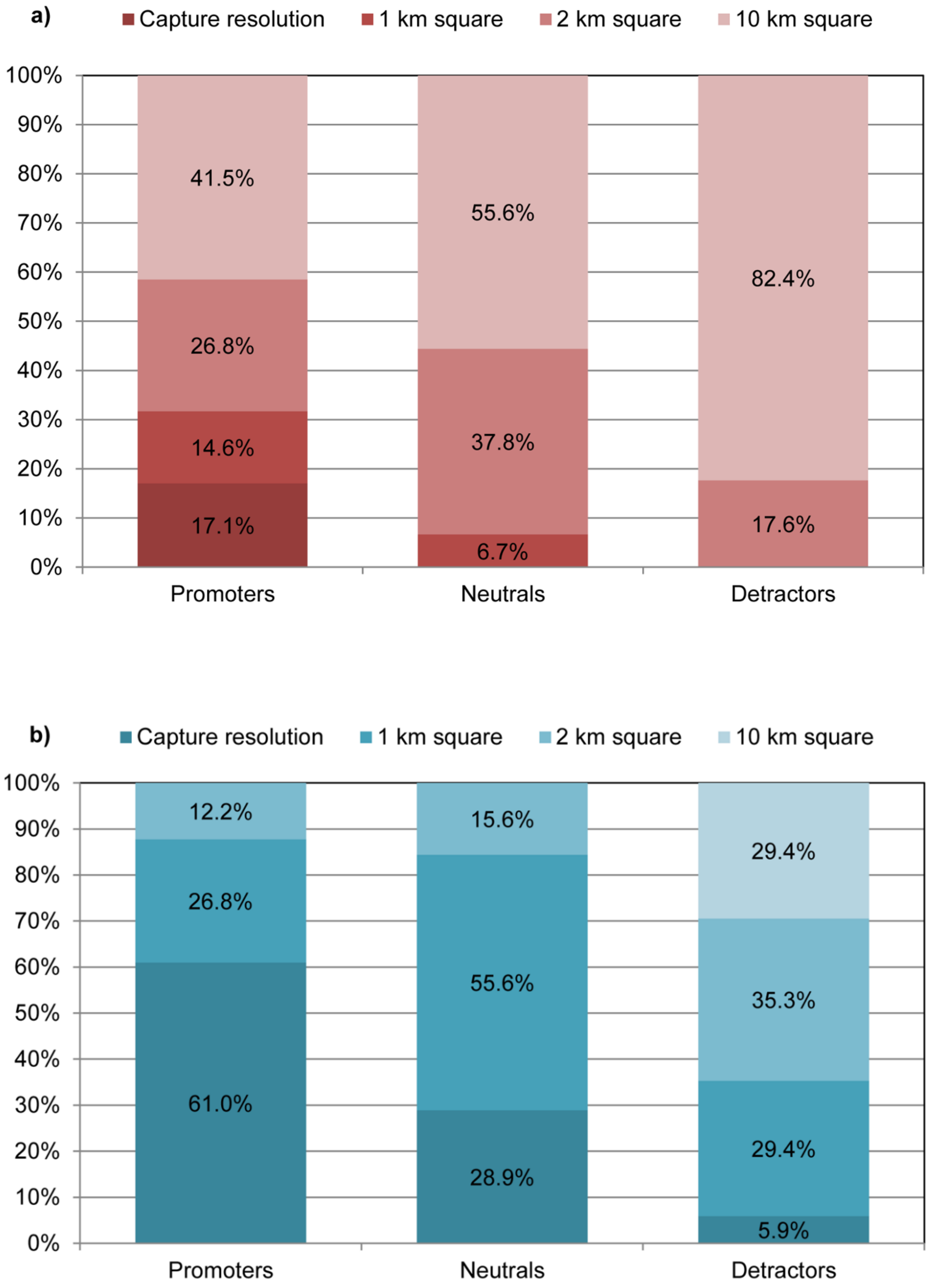

**Fig. A6.2** Preferred resolution of open access records of a) threatened species and b) widespread species among regional co-ordinators in different modified NPS categories.