Predicting the ecological impacts of species on the move with climate change in the UK

PhD in Environmental Science

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Declaration of Original Authorship

I confirm that this is my own work and the use of all material from other sources has been properly and fully acknowledged.

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Abstract

Species redistribution due to climate change poses significant challenges for biodiversity conservation. Investigation of species' shifts requires biological records; however, we have limited information on how different sources vary and whether novel approaches which employ data from social media offer any new insights. While Species Distribution Models (SDMs) are often employed to project species' distributions under the assumption of niche conservatism, that a species' niche will remain broadly consistent over time, this assumption has rarely been tested during periods of climate change. Although the environmental consequences of species redistribution are recognised, there is no established framework to assess these impacts; frameworks from invasion biology could provide some insight, however, this has never been tested in practice. This research addresses these key areas to provide crucial advancements to the field: to improve our understanding of the complementarity between different types of biological recording, test whether species consistently track climate changes to maintain their niche and evaluate the applicability of invasion frameworks for assessing impacts of climate-tracking species. A case study of the Banded Demoiselle (*Calopteryx splendens*) emphasises the importance of integrating diverse data types when determining species changing distributions, with observations from Facebook, Twitter, and Flickr broadening the geographic scope of occurrences and more fully capturing species' habitat requirements. Several Odonata (dragonflies and damselflies) exhibited temporal inconsistencies in their realised niche during a period of recent climate change, contradicting assumptions of niche conservatism with important implications for the application of SDMs. Finally, the applicability of invasion frameworks for climate-tracking species is demonstrated, revealing both negative and positive ecological impacts of Odonata and Hymenoptera (wasps, bees, ants, and sawflies) species shifting due to climate change. These findings provide insight for conservation and management which must adapt to the uncertainty of species future distributions and the associated ecological impacts

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1. Introduction & Context

1.1. Introduction to climate change

The severe and imminent threat to the human and natural world of anthropogenically driven climatic change is widely recognised and thoroughly investigated. Climate change can be defined as any identified and observable change in the state of any climatic property that persists for an extended period – as is described by the Intergovernmental Panel on Climate Change (IPCC; Matthews, 2018). This definition therefore encompasses climate change that occurs due to natural internal processes as well as natural external forcings such as volcanic eruptions and solar cycles (Matthews, 2018). Conversely, climate change is typically used to convey anthropogenic climate change, such as by the United Nations Framework Convention on Climate Change (UNFCCC) who specify that climate change is defined as that which is directly or indirectly attributed to human activity, with climatic change occurring naturally, defined separately as climate variability (United Nations, 1992). For ease, here the term climate change will be used similarly to the UNFCCC to denote climate change which arises from anthropogenic changes to the composition of the atmosphere and biosphere.

Climate change research is a unique field of study as it involves a structured and coordinated scientific effort that is unparalleled within any other discipline. Climatic changes have been widely and extensively detected, with rapid mean global atmosphere, land, and ocean warming trends clearly distinguishable from natural climatic variability. Furthermore, despite continued public controversy and debate (Van Houtan et al., 2021), the IPCC (2023) reported that increasing greenhouse gas concentrations and resulting warming is unequivocally due to human influence. Our knowledge of climate change relies on global, long-term observations that consist of direct conventional and remotely sensed measurements, as well as proxy observations and palaeoclimatology. Such observations are central to characterising and detecting climate change, determining rates of change, as well as for developing and evaluating climate models (Simpkins, 2017). Climate modelling is also crucial to our investigation of climate change in terms of providing idealised experiments for process understanding, exposing the impact of anthropogenic forcing and attributing warming to human impact, as well as simulating and interpreting past and future changes in climate (Simpkins, 2017).

Such methods have provided a comprehensive picture of long-term climatic change that reveals the warming of our land and ocean surface temperature and impacts across the Earth's spheres. The clearest impacts for the cryosphere include global retreat of glaciers, decrease in Arctic Sea ice, surface melting of the Greenland ice sheet, and decrease in Northern Hemisphere spring snow cover (IPCC, 2023). Major impacts for our oceans are increasing global sea level as a result of ice loss on land and thermal expansion from ocean warming, as well as the chemical effects of ocean acidification and deoxygenation driven by human emissions (IPCC, 2023). Climate warming is also very likely to result in weakening of the Atlantic Meridional Overturning Circulation with the IPCC (2021) concluding that abrupt circulation changes, although unlikely cannot be ruled out along with the resulting abrupt shifts in regional weather patterns and water cycling that would occur as a result. In addition, climate change is affecting weather and climate extremes across the globe with increases in the frequency and intensity of hydrological droughts, heavy precipitation, and hot extremes, including heatwaves, in some regions (IPCC, 2023). Consequences for the biosphere include polewards shifts of the climatic zones in both hemispheres and lengthening in the growing season in the Northern Hemisphere mid latitudes (IPCC, 2021).

With vast attention and scientific investigation, the broad-scale principles of climate change are well-researched, however, specific details are less well understood, and many uncertainties remain (Simpkins, 2017). Future climate will depend on future emissions that in turn are determined by socio-economic conditions, policy, as well as the sensitivity of the climate system. Modelling future dynamics is therefore based on plausible scenarios designed to take into account a range of possible eventualities in order to deal with the uncertainty in future emissions and development (Van Vuuren et al., 2011). The response of the climate to such scenarios depends on climate sensitivity and feedbacks, some of which are more comprehensively understood, whereas others such as the cloud feedback are complicated and determined by several competing elements (Ceppi et al., 2017). Continued research and developments of climate modelling are gradually improving our understanding and ability to predict future change (Simpkins, 2017); however, it remains difficult to capture the complexity and intricacies of the climate system, particularly as human driven climate change occurs alongside natural variability across a range of timescales and patterns are heterogeneous across time and space. Climate change is therefore a continually developing and extremely relevant research topic, where investigating the details will be crucial for enhancing our resilience against future change.

1.2. The rise of redistribution science

The natural world has been greatly altered by human activity across the globe causing a rapid decline in biodiversity and threatening species with global extinction at a rate unparalleled in the past 10 million years (Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services [IPBES], 2019). This biodiversity crisis threatens human civilisation as nature is essential for human survival and well-being (Singh, 2002; IPBES, 2019). Many human drivers endanger ecosystems such as habitat change, direct species exploitation, pollution, and invasive species, as well as climate change which poses a direct risk as well as exacerbating the impact of other drivers due to compounding effects (IPBES, 2019). The biodiversity and climate crisis are inherently linked as climate change alters the biosphere which in turn impacts the climate through modifications to greenhouse gas concentrations in the atmosphere and changes to the carbon cycle. Investigations at the interface between the environmental issues of biodiversity loss and climate change are essential as both threaten the integrity of the Earth and its ability to sustain human life (Legagneux et al., 2018).

It is widely acknowledged that anthropogenically driven climate change poses stark threats to biodiversity with impacts documented across every ecosystem on Earth (Scheffers et al., 2016). Scheffers et al. (2016) provide a comprehensive overview of such impacts across different levels of biological organisation including reduced genetic diversity and body size at an organism level, changing range location and habitat quality at a species level, altered phenology and dynamics on a population level, and modified biomass and primary productivity on a community level. Such impacts have unprecedented consequences for human systems (Van Houtan et al., 2021) such as threatening food security through reduced agricultural yield (Muluneh, 2021) and human health through changing distributions of pathogens (Scheffers et al., 2016); the impact of climate change on biodiversity impacts ecosystem functioning and the provision of ecosystem services and resources for human-use (Scheffers et al., 2016).

Of these vast impacts of climate change, one of the most rapid across the marine, freshwater, and terrestrial realms is the resulting movement of species to track their optimal habitat conditions (Scheffers et al., 2016). Such species redistribution as a consequence of contemporary climate change is a recent, yet rapidly developing field of research, with a proliferation of research on the topic within the past decade (Bonebrake et al., 2017). As such these species 'distribution changes' or 'range shifts' have been extensively reported within literature across a variety of environments for a range of taxa (Pecl et al., 2017), however coverage remains geographically, taxonomically, and methodologically incomplete (Bonebrake et al., 2017). As examples of climate-driven movements by species accumulate, species redistribution ecology is rapidly emerging as a distinct field of research, a term used by Bonebrake et al. (2017) to include the study of species movements and the resulting impact for ecosystems and societies.

The ability of species to shift their distributions is one of several mechanisms by which species are able to survive a rapidly changing climate (Mason et al., 2015), and is considered a key aspect of species persistence on both regional and global scales (Wallingford et al., 2020). Such distribution changes include species shifting at their range margins to expand into new areas and retract from others, as well as changes of species' abundance within their current geographic range (Bowler et al., 2015). This leads to the arrival of new species and the extirpation of others within any given location, with the potential for colonisation and extinction of species on national and international levels

(Bennie et al., 2013). Climate change can also result in changes to species migratory routes such as due to shifting of their breeding and non-breeding ranges (Howard et al., 2018). The term 'species on the move' has been used to broadly describe any species movements that can be attributed to climate change (Pettorelli et al., 2019) – a connection that is exceedingly difficult to define and establish (Chen et al., 2011a; Pearce-Higgins et al., 2017).

Species redistribution ecology is a rapidly developing discipline, driven largely by the anticipated wide-reaching and severe consequences of species on the move for ecosystems and the societies they support (Twiname et al., 2020; Wallingford et al., 2020). With such implications having already been observed, this reshuffle of biodiversity is an imminent issue and not a concern for the future, as considered until late (Bonebrake et al., 2017). Nevertheless, despite widespread and numerous reports of climate-driven species redistributions, studies into the resulting impacts are lacking (Wallingford et al., 2020).

To adequately predict the consequences of climate change for biodiversity, systems need to be put in place for the rapid detection of species on the move and assessment of their potential positive and negative impacts. Devising such systems proves difficult as redistribution science traverses many established disciplines with complex processes that occur across different biological levels and spatiotemporal scales (Pearce-Higgins et al., 2017; Twiname et al., 2020). Consequently, the next steps for this emerging field will have crucial implications for conservation requiring a clear direction and interdisciplinary approach (Bonebrake et al., 2017; Twiname et al., 2020).

1.3. Thesis rationale and structure

This introduction has set the context for this thesis by introducing climate change, outlining the resulting movement of species, and providing a background to the growing field of species redistribution. This introduction is followed by a literature review which provides a detailed summary of current knowledge on this topic, firstly summarising published literature regarding species on the move, secondly outlining the data requirements and methods employed to track shifting species, and thirdly describing impacts of species on the move and potential approaches to assessing these impacts. The literature review is concluded by outlining the three main research gaps this thesis seeks to address, and the subsequent section highlights the key objectives of this research and the aims and hypotheses for each data chapter. The methodology introduces the study system and region, as well as describing the methods applied, including the data employed, techniques to measure geographic distribution change, species distribution modelling, climate niche analysis, and the statistical methods adopted.

This thesis is comprised of three data chapters, the first two have been published in peer-reviewed journals and the text in these chapters have been retained as published, however, they have been formatted for consistency with the rest of this document. The three data chapters are as follows:

Chapter 5: Investigating the potential of social media and citizen science data to track changes in species' distributions:- This chapter provides an assessment of biological records to determine the complementarity between different types. *Published in: Ecology and Evolution.*

Chapter 6: Investigating odonates' response to climate change in Great Britain: A tale of two strategies:- This chapter undergoes a simultaneous investigation of Odonates' (dragonflies and damselflies) geographic distribution change and temporal climate niche dynamics during a recent period of climate change. *Published in: Diversity and Distributions.*

Chapter 7: Insight from invasion biology for assessing the impacts climate-tracking Odonata and Hymenoptera species:- This chapter provides a semi-quantitative, spatial assessment of the potential consequences of climate-tracking Odonata and Hymenoptera (wasps, bees, ants, and sawflies) species by adopting impact assessment frameworks utilised within invasion biology. *In preparation for publication.*

This thesis culminates in a discussion covering the key contributions to the field, the main limitations of the employed methodologies, and the implications and challenges that remain for conservation efforts. This is followed by a concluding section that provides a comprehensive summary and final remarks.

1.3.1. Contribution to published work

Chapters 5 (*published in Ecology and Evolution*) and **6** (*published in Diversity and Distributions*) of this thesis constitute published work in which I was the first author, however, as the content of these articles have been retained within my thesis as published, I will detail here the contribution made by all other co-authors. N. Pettorrelli as well as my other supervisors (J. Neumann, L. Shaffrey, C. Cheffings, and K. Norris) aided in the conceptualisation of these analyses. In revising my manuscript drafts, N. Pettorelli also contributed to the writing of these papers, and all supervisors aided in reviewing final manuscript drafts. H. Häkkinen also provided comments on both manuscripts and is a co-author for the paper in *Ecology and Evolution* (**chapter 5**). Other than the contribution by H. Häkkinen specifically detailed below, I conducted all the data collection, investigation, results, and analyses for these papers.

During peer review of the *Ecology and Evolution* (**chapter 5**) article, amendments were suggested to account for recorder effort when implementing Species Distribution Models (SDMs). H. Häkkinen accessed proxies for recorder effort, revised my original model code to incorporate these into SDMs to account for recorder bias, and subsequently generated new output and graphics from these new models. H. Häkkinen also undertook a sensitivity analysis to review the effect of differing levels of precision for the social media data which I collected. For my final research chapter (**chapter 7**), I implemented this 'bias covariate correction' method for my SDMs, where I accessed and processed similar recorder effort proxy variables and implemented bias corrections within my own models. With this in mind, to ensure that the work I present here is my own, I have only incorporated the methods which I applied myself into this thesis Methodology (**chapter 4**), therefore any reference to 'bias covariate correction' within this section refers to the analysis conducted for **chapter 7**. The figures included in **chapter 5**, are the ones in which I created following my original model development, and therefore differ slightly from the figures in the published version in *Ecology and Evolution* created by H. Häkkinen. Furthermore, reference to accounting for recorder effort within **chapter 5**, refers to the work conducted by H. Häkkinen and the associated supplementary documents which are available alongside the publication are excluded from this thesis.

2. Literature Review

2.1. Current knowledge of species on the move

With the rise of redistribution science, changes in species distributions as a response to climate change is becoming a widely recognised and researched phenomenon (Hickling et al., 2006; Scheffers et al., 2016). Evidence of species tracking their suitable ecological niche across the landscape is rapidly building with cases reported for a wide range of taxa across numerous different environments. Despite this, the global picture of this biodiversity reshuffling according to climate change is far from complete with certain geographic dimensions of species shifts, distribution parameters and climate change variables remaining understudied (Lenoir & Svenning, 2015).

The most extensively documented climate change driven species redistribution changes are latitudinal and elevational shifts (Lenoir & Svenning, 2015) due to the widely accepted positive spatial autocorrelation of temperature across these gradients and the resulting directional impact on species range shifts (Guo et al., 2018). As such, species can be predicted to shift towards the poles and to higher elevations to follow broad geographic shifts of isotherms over time amidst a warming climate (Burrows et al., 2011; Guo et al., 2018). As data builds, researchers such as Chen et al. (2011a) have sought to summarise findings to determine how rapidly species are responding to contemporary climate change and whether the distribution changes reported are sufficient to track temperature changes. According to their meta-analysis across terrestrial organisms they reported median shifts to higher elevations and higher latitudes at a rate of 11.0 m and 16.9 km per decade, respectively. A more recent review of global redistribution rates for plants and animals across terrestrial, freshwater, and marine ecosystems by Rubenstein et al. (2023), reported relatively comparable average shifts of 9 m and 11.8 km per decade towards higher elevations and latitudes, respectively.

Theoretically, however, distribution shifts can occur over different geographical dimensions as species react to changing gradients of environmental parameters, including latitude and longitude as well as elevation for terrestrial species and depth for marine (Guo et al., 2018) – dimensions that are not equally investigated and reported within literature (Lenoir & Svenning, 2015). As redistribution science has developed, the number of scientific reports documenting other shifts has increased (Lenoir & Svenning, 2015) with evidence of shifts now spanning a range of directions (Pinsky et al., 2013). This includes reports of longitudinal shifts (Keith et al., 2011; Mattila et al., 2011; Van der Wal et al., 2013; Wu, 2016) as well as shifts that occur, perhaps more counterintuitively, towards lower latitudes and elevations (Lenoir & Svenning, 2015; Bonebrake et al., 2017). For the numerous studies that report average shifts towards higher latitudes, elevations, and greater depths, several individual species across a variety of taxa have been found to be static or shifting in the opposite direction to that typically expected (Lenoir et al., 2010). Rubenstein et al. (2023) performed a systematic review of global redistribution studies and discovered that in fact less than half of all range-shift observations (46.60%) were consistent with common hypotheses – i.e. towards higher latitudes, higher elevations, and greater marine depths. Such variation in empirical evidence has been attributed to species-specific climate tolerances and characteristics (Lenoir & Svenning, 2015), other constraints such as land-use changes, habitat modification and physical barriers (Keith et al., 2011; Bonebrake et al., 2017) and indirect biotic factors (Lenoir et al., 2010). Contrarily, such atypical shifts have often been found to be consistent with climate change when investigated at a fine enough spatial resolution (Engelhard et al., 2011; Lenoir & Svenning, 2015; Bonebrake et al., 2017). Despite such developments, multidimensional studies of species range shifts across all geographic dimensions simultaneously, remain lacking (Lenoir & Svenning, 2015).

Studies into species distribution changes according to climate change have largely been concerned with species range shifts at their margins and expansion at a species leading-edge into newly suitable climatic areas. Reports of species contractions at their trailing edge are less conclusive (Chen et al., 2011b); this has been theorised to be due to species persistence within favourable microclimates (Suggitt et al., 2018) as well as a methodological artefact due to the difficulty in detecting species extirpations from species occurrences at the coarse resolutions typically applied (Thomas et al., 2006; Massimino et al., 2015). Furthermore, abundance patterns are also less well-studied including changes in the abundance of warm- and cold- adapted species (Bowler et al., 2015), as well as shifts in species' abundance within their geographic range (Huang et al., 2017). This is largely due to the species occurrence data required for such analyses that are limited to specific, well-recorded taxa (Outhwaite et al., 2018).

Distributional studies – which primarily investigate latitudinal and elevational shifts – are largely focussed on changes in temperature and often ignore other important climatic variables (Van der Wal et al., 2013). This temperature centric approach is unsurprising as rising temperature is one of the most direct, noticeable, and widely acknowledged aspect of climate change (Spence & Tingley, 2020), however, such approaches ignore the complex regional patterns of climatic changes that shape resulting species distribution change (Lenoir & Svenning, 2015; Bonebrake et al., 2017). Moving away from this temperature-based approach, substantial climate change research has also investigated the role of changing water availability and precipitation (Spence & Tingley, 2020). Such developments are enhancing our understanding of observed range shifts as well as improving model performance when predicting current and future species distributions (Spence & Tingley, 2020).

Following such developments, rather than to assume a poleward or elevational shift solely due to temperature increases, it has been suggested that it would be more beneficial to consider local climate velocity, that is to determine the direction and speed that a species at a particular point would need to move under climate change to maintain their climatic niche based on the environmental variables and time frame considered. Using this approach, Burrows et al. (2011) described the complex mosaic of global shifts theoretically required by species to track temperature during 50 years of past climate warming and found these projections deviate from simplified polewards shifts previously predicted and detected. This theoretical approach has also been applied to specific taxa such as by Van der Wal et al. (2013) who determined that the climate velocities of distribution change required by Australian birds due to climate change were greater when considering both temperature and precipitation than when only considering a latitudinal shift due to temperature alone. This suggests that previous predictions of shifts under climate change with a unidirectional and univariate focus may significantly underestimate the shifts that could occur (Lenoir & Svenning, 2015; Huang et al., 2017).

Other weather variables in addition to temperature and precipitation, affect organismal performance and therefore shape species movement under climate change (Berggren et al., 2009). When conducting multidirectional studies, species distribution shifts have been found to be determined by interactions between multiple climate factors and unattributable to any single climatic variable, as was concluded by Gillings et al. (2015) for birds in Great Britain. Furthermore, different climatic variables can operate along different spatial trajectories, driving species in different contradicting directions making it difficult to evaluate past distribution shifts and project future ones (Gillings et al., 2015). For example, air and ocean currents influence dispersal of marine and terrestrial organisms; these currents and can act to either hinder or facilitate species shifts depending on the extent to which their spatial gradients covary with thermal gradients (Molinos et al., 2017; Spence & Tingley, 2020). As such, species widely reported inability to perfectly track temperature changes across the landscape may largely be due to the role of other abiotic variables in limiting species' range (Spence & Tingley 2020), rather than a lag that is often explain by dispersal limitation or biotic interactions (Littlefield et al., 2019).

In summary, current research within redistribution ecology are restricted in terms of geographic dimensions, parameters, and abiotic variables investigated. This emphasises the need for further research into species distribution shifts that deviate from temperature-based polewards and elevational shifts. Instead, it has been recommended that studies consider directionality in climate and distribution changes (Gillings et al., 2015; Huang et al., 2017). In addition, further research is required to determine the relative influence of different climatic variables, and the metrics used to capture them, on distribution shifts (Garcia et al., 2014; Littlefield et al., 2019), as well as the effects of climate averages versus climate extremes (Huang et al., 2017), and the influence of extreme events (Littlefield et al., 2019).

2.2. Tracking species on the move

2.2.1. Species occurrence data

A biological record is simply a log of a species at a particular place at a certain time (Isaac & Pocock, 2015; Pocock et al., 2015). When collated into a dataset, biological records are an invaluable monitoring tool that can be used to address large-scale questions about biodiversity and environmental change as they provide information both with a high spatial resolution as well as a wide spatial extent owing to the extensive geographic spread of contributing recorders (Pocock et al., 2015). Biological records originate from a variety of sources and are coordinated by a number of different monitoring schemes and societies. These data-holding institutions can make their records accessible via online portals such as the Global Biodiversity Information Facility (GBIF) which hosts biological records from institutions worldwide (Powney & Isaac, 2015), often being fed directly from national databases such as the National Biodiversity Network (NBN) Atlas in the UK.

Biological recording encompasses a range of recording activities that can be broadly categorised into different types. These categories are useful for describing the varying collection of biological records and resulting information content; however, they are by no means exhaustive of all species occurrence records, nor is each type fully distinct from one another, with various biological records falling between these types.

For a small number of taxa in a few countries, structured monitoring schemes exist to provide systematic and focussed recording (Isaac et al., 2014). Such schemes require dedicated participants and typically involve standardised protocols and therefore provide high-quality information such as annual abundance records that can be utilised to provide robust estimates of trends (Isaac et al., 2014; Pocock et al., 2015). Such high-level and coordinated monitoring is only possible for a select group of taxa, namely those that are the most charismatic with widespread interest from volunteer recorders, scientists, and naturalists. For example, for birds, the Breeding Birds Survey (Field & Gregory, 1999) and the Seabird Monitoring Programme (Walsh et al., 1995) in the UK, and the North American Breeding Bird Survey (Sauer et al., 1997). There are also several national butterfly monitoring schemes which have proliferated in Europe following the establishment of the UK Butterfly Monitoring Scheme in 1972, with Butterfly Conservation Europe founded in 2004 to coordinate butterfly monitoring efforts across Europe (Van Swaay et al., 2008).

Most biological recording however fits within opportunistic, unstructured recording schemes. These are generally coordinated by individual specialist recording schemes or societies that collate records with a particular taxonomic focus (Pocock et al., 2015). For such schemes there is no formal procedure in place for the collection of records, therefore observations are predominantly collected and submitted by volunteer recorders at their own discretion. Unlike focussed schemes with set procedures, observations from unstructured schemes vary greatly in terms of their quality depending on how such observations are collected, validated, and stored. Due to variation in recorder activity these records also have inherent sampling biases that must be overcome by researchers to extract useful information for different uses (Isaac et al., 2014). Nevertheless, biological records of this type are high in quantity and are taxonomically varied providing a wealth of information unavailable from structured monitoring programmes.

The content of biological records is frequently disseminated through published atlases with different levels of organisation and information content. In some cases, these atlases contain opportunistic data that has simply been collated together over a specific period, in other cases these atlases are the result of more focussed recording where a concerted effort is undertaken to fill in any geographical gaps left by unstructured schemes (Pocock et al., 2015). For these atlases, an attempt is made to ensure the whole region is recorded with adequate coverage over a limited amount of time with a degree of organisation and focus required (Isaac et al., 2014). As such, atlases can be considered a separate category of biological recording, with methodologies varying between more structured, managed schemes to unstructured, opportunistic ones. For certain taxa, where more than one atlas has been published, atlases are particularly useful tools for researchers for assessing changes in species distribution between atlas periods (Isaac et al., 2014).

It has also been acknowledged that with technological advancements making it easier to submit records, biological recording has become more accessible to the general public with a wider participation in recording forming another category of biological records as specified by Pocock et al. (2015). This category would include citizen science, a rapidly growing research practice where the public is directly involved in the scientific research process (Cooper et al., 2014). In the case of biological recording, citizen science projects offer invaluable contributions of species occurrence records as well as opportunities to engage people with nature and science (Pocock et al., 2015). For example, the adoption of a mobile application by the UK Ladybird Scheme enabled a rapid increase in recorders (Pocock et al., 2015). Several citizen science applications now exist for biological recoding, such as the iNaturalist global platform that enables individuals to photograph organisms along with a location and date, which once identified through the global community of naturalists, forms a biological record (Nugent, 2018). More focused citizen science projects also exist, such as the Garden

BirdWatch run by the British Trust for Ornithology that encourages the public to collect bird observations in their own gardens. This programme offers somewhat structured recording as recording periods are specified with participants encouraged to repeat recording at the same time and place for each recording session (Catlin-Groves, 2012). Furthermore, it should be noted that as the vast majority of biological records are collected by volunteer recorders, wider participation covers much biological recording (Powney & Isaac, 2015).

Biological records differ in their format and information content; numerous species observations are collated through unstructured recording schemes and citizen science applications that typically provide 'presence-only' data in that they are incidental records of only the species that were observed at a particular point in space and time (Isaac & Pocock, 2015). The difficulty with such data is that little is known about the intervening space between occurrence records as there is no way to conclude whether such areas reflect visits where a particular species was not encountered, where no recorders visited, or simply a recorder only noted particular species which they deemed interesting (Pocock et al., 2015). This is as opposed to presence-absence data resulting from more structured monitoring schemes where set species are surveyed for within particular geographically defined areas and those present are noted. Although presence-absence data is often considered optimal, developments have led to novel methods of implementing presence-only records for different applications (Pearce & Boyce, 2005; Tsoar et al., 2007; Valavi et al., 2021).

(a) The 'recorder effort problem'

Biological records provide essential ecological information; however, they are associated with several different issues and uncertainties depending on the sampling method utilised. Different types of biological recording have different attributes, covering different spatial and temporal extents as well as being measured at different resolutions. In addition, occurrence records vary in their information content according to whether they are presence-only, presence-absence or abundance records and the extent to which additional information is provided about the observation process (Altwegg & Nichols, 2019). These features must be considered when conducting ecological research, particularly when collating data from different sources.

In determining a species distribution, presence-absence data is typically considered the highest quality, especially when information about the sampling process is included (Altwegg & Nichols, 2019). Such data is typically restricted to records from structured monitoring programs as well as atlases where a concerted effort is made to report all species encountered within a particular specified area (Altwegg & Nichols, 2019). The primary issue for data of this type is the extent to which a species has been reliably detected due to imperfect detection (Kissling et al., 2017). With detection probability typically less than 70% for common species, false negatives are an important source of error, whereby the presence of a species that occurs is not reported; an issue that is even more prevalent for uncommon or secretive species (Altwegg & Nichols, 2019). Another detection error that can occur is the misidentification of a species leading to false positives, where a species is reported as present at a location where it does not occur (Altwegg & Nichols, 2019).

The vast bulk of biological records are the result of opportunistic surveys and citizen science projects that typically provide presence-only data. As well as being subject to species misidentification and detection error as with presence-absence records, their unstructured collection leads to variation in recorder activity and associated sampling biases. These biases create statistical problems for researchers and limit the scope of scientific applications of these records – an issue that has been termed the ‘recorder effort problem’ (Isaac & Pocock, 2015). The opportunistic collection of records results in uneven sampling over both space and time, with species distributions and environmental gradients commonly sampled in an unrepresentative way, obscuring conclusions drawn when these biases are unaccounted for (Altwegg & Nichols, 2019). Generally, recording intensity has increased over time with an approximately exponential growth of records for many schemes (Isaac & Pocock, 2015). This leads to problems when interpreting records, as perceived expansions in species’ range may be an artefact of increased recording intensity over time. Through space, records are aggregated where recorders live or tend to visit, with spatial coverage of observations being patchy for most taxa (Altwegg & Nichols, 2019).

Another bias with biological records is uneven sampling effort per visit, which is the extent to which a set of records collected at a particular time and location reflects the total species that were present and recordable (Isaac & Pocock, 2015). This depends on the intensity of the search and the set of species that were surveyed, as well as each species’ detectability. This phenomenon is difficult to investigate as most biological records are incidental reports of particular species rather than the results of surveys noting all species present. A direct estimate of survey effort more generally is lacking for many biological surveys and atlases and so proxies must be used to allow an estimation of survey effort so that it is accounted for before any conclusions can be drawn regarding changes in species distributions (Kujala et al., 2013). As well as influencing sampling effort, species detectability is also uneven throughout space and time and influenced by recorders and their methods causing another well-known bias with biological records (Isaac & Pocock, 2015).

More broadly, there are other uncertainties across all types of biological records, such as in the precision and accuracy of coordinates (Kissling et al., 2017), with errors in georeferencing commonly reported (Yesson et al., 2007), as well as temporal uncertainty in terms of precision of the collection time reported (Kissling et al., 2017). Furthermore, there are biased representation of taxa across records (Pettorelli et al., 2019) and taxonomic uncertainty in terms of species identification and ambiguous scientific names (Kissling et al., 2017).

(b) The potential of social media

Social media offers a novel source of information for answering ecological questions about biodiversity, species distributions, and the impacts of climate change. Social media websites and applications allow users to post content of any kind, offering vast amounts of untapped, freely available information when this content is relevant to the ecological questions being investigated. For example, if an individual takes a picture of a species and posts this online, this could constitute a biological record providing additional information such as a location and date are also available with the picture providing verification of species identification. As such, scientists are beginning to recognise the potential of social media information as a source of scientific information, with a few select cases where species occurrence records have been gathered from social media to investigate species distributions. This concept has been tested using geo-tagged Flickr images for the snowy owl and monarch butterfly (Barve, 2014), for two pollinator species and two flowering plants in Australia (ElQadi et al., 2017), as well as for cetacean species in the Mediterranean using photos and videos posted on a variety of social media platforms (Pace et al., 2019). Social media information has also been implemented to detect non-native distributions such as for the oak processionary moth, emerald ash borer and Eastern grey squirrel impacting forest ecosystems (Daume, 2016) and freshwater turtles in the UK (Allain, 2019).

Social media illustrates another example of public engagement with science. Citizen-science networks such as iNaturalist can be considered a form of social media, in that they provide individuals with a platform and profile to share content they desire. In addition, involvement of the public in science is ever more feasible for scientists where existing online social media applications and websites can be utilised for collecting biological records, for example by encouraging the use of specific hashtags on Twitter or Instagram or creating specific Facebook pages to provide a platform for individuals to add content for a specific scientific purpose.

As the use of social media advances, a new field of research is developing that acknowledges such sources of information that has been termed by Jarić et al. (2020) ‘internet ecology’ or iEcology. Jarić et al. (2020) define iEcology as the implementation of online data for ecological research where the digital data used has been collected or uploaded by users for a different purpose. As well as the data itself in the form of text, images, or videos, iEcology also encompasses analyses of online activity such as search engine uses (Jarić et al., 2020). iEcology has the potential to provide insight into many research areas, opportunistically utilising information available online for a new purpose, with Jarić et al. (2020) citing the most common application being the exploration of species occurrences and spatiotemporal trends. Social media information can therefore constitute iEcology, in addition to social media data that has been managed and designed for answering specific scientific questions.

Social media data has the potential to complement existing sources of biological data for monitoring species on the move. For traditional data, information is collated and verified before it is made available for scientific research, whereas social media data is uploaded instantly providing a rapid source of species occurrence information. Social media also has the potential to capture species occurring in locations missed by more traditional records and therefore could capture a more complete understanding of a species ecological niche. For example, when overlaying Flickr and traditional occurrences of honeybees in Australia, ElQadi et al. (2017) found that social media data extended species' distributions closer to urban centres as well as tourist destinations in remote locations, two contrasting areas that may be underrepresented from traditional biological records. Social media data is particularly useful when other biological records are lacking, such as for Allain (2019) who found a greater number of occurrence records from Flickr for non-native freshwater turtles in the UK than available on Record Pool, the UK's herpetofauna recording scheme. In addition, Allain (2019) also found Flickr data to be more consistent over time than traditional records.

It is expected that the implementation of social media, including iEcology, will continue to develop rapidly, along with increased application of such information for determining species distributions (Jarić et al., 2020). The utility of social media information for monitoring species on the move is an interesting and currently understudied prospect. The potential of social media to this end has been demonstrated, such as by Pettorelli et al. (2019) who identified 10 species in new locations in the UK using keywords and hashtags on social media. Advances following a similar approach could aid in our understanding of species movements due to climate change.

2.2.2. Mapping species' distribution

There are numerous different ways in which scientists have attempted to determine a species' distribution. Gaston and Fuller (2009) classified these into 5 broad categories of methods – marginal occurrences, range wide occurrences, habitat distributions, statistical modelling, and process-based modelling. With the first two approaches, a species distribution is determined from species occurrence records alone, without the use of additional environmental information. For marginal occurrences, the boundary of a species' distribution is inferred from the outermost occurrences of species and interpolation between these points (Gaston & Fuller, 2009). If occurrence records are of a sufficient density, then the distribution can be determined from these records without any additional statistical methods, which Gaston and Fuller (2009) described as range wide occurrences. With such approaches, records are often grouped into grids as with many distribution atlases (Gaston & Fuller, 2009).

In determining a species distribution on the basis of occurrence records, these methods must first account for the presence-only nature of the data, as the failure to record a species at a particular location and time does not necessarily indicate the absence of this species. Biases due to the recording process must also be overcome, particularly in ensuring that the growing quantity of biological recording over time does not indicate a false signal of species expansion with climate change. To this end many different methods have been utilised including sub-setting of records to include data of a similar level of recording intensity over time and space, or to separate out the most unbiased data based on some criteria of data quality, and aggregating records into groups such that the spatio-temporal biases in the data will be averaged out (Isaac & Pocock, 2015). More recently statistical methods have developed to account for the data collection process such as occupancy modelling that retain more information from the original biological records and which can be implemented to estimate the probability of species occurrence from presence-only datasets accounting for imperfect species detection (Altwegg & Nichols, 2019). Although typically reserved for taxa that are the most well-recorded, as these statistical methods have become more sophisticated, they are even becoming applicable for species where recording intensity has been comparatively low (Outhwaite et al., 2018).

Despite these developments, it is exceedingly difficult to utilise these biological records to detect species on the move as changes in species distribution first need to be analysed to determine whether they deviate from the known distribution of that species and then further information is required to review whether this shift can be attributed, at least in part, and with a certain degree of confidence, to climate change. Biological recording would benefit from the improvement of the information content of records, such as details regarding the survey method, in order to more effectively remove bias, as well as additional information for conservation and research purposes such as flagging species outside of their typical geographic range. Furthermore, tracking species on the move requires rapid information on a species distribution, however, opportunistic biological records must go through an essential process of data verification that can be time consuming in certain cases such as for schemes or societies with few volunteer experts and organisers that manually complete verification before making records publicly available via platforms such as GBIF (Isaac & Pocock, 2015). On the other hand, recent technological advances and the growth of citizen science within biological recording has enabled more rapid collection and dissemination of records. For example, applications, such as iNaturalist allow for instantaneous upload of occurrences as well as rapid validation crowdsourced from the platform’s internal community of naturalists and experts, before feeding directly into GBIF (Nugent, 2018).

Even once the occurrence of a species outside its known geographic range has been detected and confirmed from biological records, the attribution of such shifts to climate change is exceedingly difficult to establish. Historically redistribution ecology has focussed on unidirectional shifts of species in terms of latitude and elevation that are assumed consistent with climate change but lacking any statistical evaluation of the link between the magnitude of range shift and levels of warming (Chen et al., 2011a; Lehikoinen et al., 2013). More recent analyses have found consistencies between the distance moved by species and the level of warming experienced (Chen et al., 2011a; Menéndez et al., 2014) whilst others have found that distribution changes are more consistent with other drivers of change or not statistically different from the dynamics expected by chance (Mair et al., 2012; Taheri et al., 2020). As the complexity of different drivers of species distributions makes it difficult to establish a causal relationship between climate change and species shifts, this link is often established by determining whether shifts occur along environmental gradients and whether shifts are consistent with those predicted according to climate-change (Thomas et al., 2010; Wu, 2016).

(a) Species distribution models

The final categories of methods to determine a species distribution distinguished by Gaston and Fuller (2009) are analogous to correlative and mechanistic Species Distribution Modelling (SDM). SDMs are statistical tools used to predict the distribution of a species across time and space using environmental information (Elith & Leathwick, 2009). Correlative SDMs, involve statistically linking spatial species occurrence data to environmental conditions to infer a species range according to predicted habitat suitability (Kearney & Porter, 2009; Buckley et al., 2011). Mechanistic or process based SDMs incorporate details to describe how environmental variables influence and constrain organismal performance (Kearney & Porter, 2009; Buckley et al., 2010). Mechanistic SDMs are data intensive requiring detailed information and understanding of the physiological features that determine a species range (Buckley et al., 2011; Spicer et al., 2019). Such information is often lacking even for the most well-researched taxa (Urban et al., 2016). SDMs allow for current distributions to be predicted as well as extrapolation to be made across space and time, making them valuable tools for predicting a species response and vulnerability to environmental change (Willis et al., 2015).

SDMs have frequently been implemented to predict how species distributions will change in response to climate change (Rapacciuolo et al., 2012; Willis et al., 2015), knowledge that in turn has been widely used to inform conservation management strategies (Sinclair et al., 2010). The approach is to relate species current distribution with information regarding current climate so that future suitable habitat can be predicted under climate projections (Willis et al., 2015). Although useful, the applicability of SDMs for projecting climate change impacts has been debated due to their associated assumptions and uncertainties, with scale mismatch being an often-reported cause of error (Seo et al., 2008; Wiens et al., 2009; Sinclair et al., 2010). To assess the transferability of SDMs some researchers have used SDMs to predict distribution shifts according to recent climate change that have already occurred, an approach applied by Rapacciuolo et al. (2012) for British plants, birds, and butterflies; they concluded that although accuracy metrics showed high explanatory power of SDMs due to their ability to predict large areas retained by species, SDMs should be applied with caution as they largely failed to capture the changes in occupancy that did occur.

2.2.3. Niche conservatism

The ‘niche’ within ecology is a pervasive concept, however, it is not uniformly defined or applied (McInery & Etienne, 2012). Here, is has already been employed to evoke the concept of the climate factors in which species typically exist, the conditions which many species are geographically tracking across the landscape as the climate changes. Hutchinson (1957) introduced the concept of the fundamental niche, the entire set of conditions in which a species or population can survive and reproduce. The realised niche refers to a subsection of the fundamental niche actually occupied by a species which is restricted by biotic interactions such as the pressures of predation, competition, and disease. Mechanistic SDMs which incorporate species physiological tolerances or biotic interactions aim to quantify specific dimensions of a species’ fundamental niche, whereas correlative SDMs or ecological niche models estimate species’ realised niche based on a set of conditions across species’ occupied distributions (Peterson et al., 2015). With global change it has been increasingly important to attempt to describe species niches, particularly their climate niche – the set of climatic conditions in which a species can survive – to predict how species might respond to climate changes (Guisan et al., 2014).

Biological invasions offer a unique opportunity to study the niche dynamics of species as they colonise and expand in new environments, outside of their native distribution (Guisan et al., 2014). Such studies have led to debates about niche conservatism, an assumption underpinning much theoretical ecology (Guisan et al., 2014; Jezkova & Wiens, 2016); it is defined as the tendency of a species’ niche to remain unchanged over time or between different geographic areas (Pearman et al., 2008). The debate of niche conservatism has important implications, such as applicability of predicting species’ distributions in their invaded range, or under altered climate conditions (Guisan et al., 2014). Under niche conservatism SDMs offer appropriate tools to estimate the invasion risks of non-native species, whereas uncertainty arises if species do not retain their niche in their new range, if they instead can shift their niche (Guisan et al., 2014). Niche shifts have been extensively studied in invasive species (Lustenhouwer & Parker, 2022) with evidence existing both for and against niche conservatism during invasions (Guisan et al., 2014); there remains to be no scientific consensus as to whether invasive species broadly conserve their niche limits or regularly exhibit niche shifts during invasion (Lustenhouwer & Parker, 2022).

The concept of niche conservatism is important with regards to species on the move; species shifting their range under climate change are theorised to be tracking their fundamental niche geographically, i.e. shifting their distribution in order to maintain a stable climate niche (Ralston et al., 2017; Sillero et al., 2022). Although niche conservatism is largely contested within invasion biology, assessments of niche conservatism under recent climate change remains largely unexplored, particularly in conjunction with investigations into species geographic distribution changes (Lustenhouwer & Parker, 2022; Tirozzi et al., 2022). With species being widely reported to lag behind climate change, unable to disperse rapidly enough to keep pace with rapid climatic changes or restricted by other abiotic or biotic barriers (Jezkova & Weins, 2016; Ralston et al., 2017), shifting their climatic niche, rather than shifting geographically, could provide an alternative means by which species are able to survive rapid global change (Sillero et al., 2022). Moreover, if a species' niche changes over time rather than remaining stable, there are important consequences for the prediction of such species distributions under future climate scenarios or calculating the local climate velocity of a species' niche.

Recently, there has been some limited investigation into the niche shifts of species over time, through climate changes. For plants, Wang et al. (2022) reported that 75% taxa broadly exhibited niche conservatism during the past 18,000 years even during periods of rapid climate change, however, there is some evidence of long-term niche shifts for some plant taxa. Similarly, albeit on a smaller taxonomic and temporal scale, Lustenhouwer and Parker (2022) established evidence for the niche shift of a Mediterranean plant induced by climate change in its native range, whilst conversely showing niche conservatism within its invaded range in Australia and California. Niche divergence has also been observed for some bird species, by Tirozzi et al. (2022) in Northern Italy and Ralston et al. (2017) in North America. Further investigation is required across other taxa and contexts to fully evaluate niche dynamics under environmental change, and to provide evidence for whether species can geographically track climate change to maintain a stable niche, or whether climate change can instead induce niche shifts.

2.3. The consequence of species on the move

2.3.1. Reported impacts of species on the move

Historically investigations into species on the move have primarily been focussed on whether species are able to 'keep up' with a changing climate, motivated by the concern that biodiversity loss and extinctions will occur if species are unable to evolve or keep pace with a rapidly changing environment (Sorte et al., 2010). As these species shifts are a mechanism by which they are able to survive a warming climate, conservation efforts have focussed on facilitating species movements through connectivity or recommendations for assisted colonisations (Sorte et al., 2010; Wallingford et al., 2020). More recently, investigations have refocussed on understanding the full range of impacts from range shifting species with evidence building that species redistribution has the potential to disrupt ecosystems with knock on effects for society (Sorte et al., 2010).

(a) Ecological impacts

It has been noted that species respond differently to climate change, and those that are able to shift their distributions into new areas do so at different magnitudes and rates. Furthermore, different species movements are driven by different climatic gradients according to their specific climatic niche requirements (Van der Wal et al., 2013). These varying responses by species to climate change result in changing interactions between species whereby some interactions between species are lost as species are separated and novel interactions develop as new species are encountered (Pecl et al., 2017; Pecuchet et al., 2020). As biotic interactions are adjusted, there is potential for negative feedbacks where interactions hinder further range shift or positive feedbacks where biotic interactions facilitate further shifts leading to cascading effects (Pecl et al., 2017; Pecuchet et al., 2020). For example, range expansion by the spiny dogfish (*Squalus acanthias*) in the United States Northeast Shelf under climate change is projected to result in enhanced spatial overlap with prey further contributing to its success and continued spread (Selden et al., 2017).

Species interactions are vital for determining community composition and structure yet there is a tendency for research to focus on abiotic impacts (Wallingford & Sorte, 2019) with only a few studies assessing the community and ecosystem impacts of species on the move (Wallingford et al., 2020). Impacts at the community level can result from changing dynamics of predation, parasitism, mutualism, and competition, effects that have been recognised for the introduction of invasive species, a well-established driver of biodiversity loss (Pecl et al., 2017). Sorte et al. (2010) conducted a meta-analysis of the consequences of climate driven shifts of marine species and found that even though rates of spread were lower for range-shifting species than introduced species, the community-level effects were comparable. The changing diversity of species within entire communities is an area requiring further investigations with the complexities and effects on ecosystem functioning being inadequately understood (Lurgi et al., 2012; Pecl et al., 2017). Species redistribution due to climate change can alter the species richness at any given location (alpha diversity) as well as the community composition of species through space and time (beta diversity) and the number of species within a wider geographic region (gamma diversity; Pecl et al., 2017).

(b) Socio-economic and cultural impacts

Society depends on the provision of services and goods from natural ecosystems, however the geographic shift in the availability of such natural resources threatens the provision of a range of ecosystem services (Pecl et al., 2017). As outlined, redistributing species alter community composition and ecosystem functioning, further influencing the provision of ecosystem services. Shifts in species and systems that support agriculture and fisheries are of particular concern due to the potential to exacerbate the challenge of pursuing food security amidst a warming climate (Rice & Garcia, 2011; Myers et al., 2017). Agricultural productivity will not only be challenged by changing environmental conditions over time but also be impacted by changing distributions of weeds, pests, and pathogens (Bonebrake et al., 2017; Myers et al., 2017). As fish species shift their distributions, some fisheries will become less productive with reduced diversity and relative abundance of species whilst other areas might become more productive with new arrivals (Rice & Garcia, 2011). Meeting food demand will depend on our ability to predict and adapt to changing distributions of species worldwide to maintain and enhance our agricultural and fisheries productivity.

Species redistribution due to climate change also has the potential to impact livelihoods – this is particularly important for indigenous communities where relationships between species within their historic range are often an integral foundation of their livelihoods, cultures, and languages (Pecl et al., 2017; Melbourne-Thomas et al., 2022). Shifting species can also impact recreation and tourism both positively and negatively with the arrival and loss of particular species affecting the intrinsic and economic value of these industries (Pecl et al., 2017).

A final and notable impact of species on the move is the effect on human health. Changing distributions of disease vectors such as ticks and mosquitoes threaten human health, particularly when infectious diseases are transmitted into societies lacking immunity and appropriate healthcare infrastructure and resources (Pecl et al., 2017; Rocklöv et al., 2023).

Understanding the scope of the socio-economic and cultural consequences of species on the move relies on information regarding the changing distribution of species that we rely on as sources of ecosystem services and which support our livelihoods, food provision, and economies as well as those species that have the potential to adversely impact human health and well-being and threaten functioning of our agricultural, fisheries, tourism, and recreation industries.

2.3.2. Insight from invasion biology

Invasive species pose a widely recognised threat to global biodiversity and native wildlife and several frameworks have been put in place to detect and evaluate the consequence of new species arriving in new environments that have the potential to cause harm (Pettorelli et al., 2019; Wallingford et al., 2020). Although developed for species that have been introduced by humans, these frameworks are also largely applicable to species that are spreading into new environments due to recent anthropogenically driven climate change. Accordingly, it has been suggested within literature that there is the potential to adjust the focus of invasion biology to evaluate the impacts of climate driven range shifts, providing an interesting and understudied concept that could fill some of the current knowledge gaps regarding the impact assessment of species on the move (Gilman et al., 2010; Sorte et al., 2010; Wallingford et al., 2020).

Catford et al. (2009) created an integrated framework for invasion biology to fit the main hypotheses at the time into one paradigm. Although based on plant invasion ecology, this framework provides a useful approach to assessing and summarising invasion risk. Catford et al. (2009) grouped factors influencing invasion success into three broad categories – propagule pressure, abiotic characteristics, and biotic characteristics which Wallingford et al. (2020) theorised might also be applicable for determining the impacts of range-shifting species. For species on the move, propagule pressure instead can be considered as dispersal pressure with characteristics such as high fecundity and wide dispersal increasing success as opposed to low fecundity and limited dispersal (Wallingford et al., 2020). Abiotic characteristics are the physical characteristics of the recipient ecosystem that increase susceptibility to invasion such as a history of disturbance or environmental stress (Catford et al., 2009). Finally, the biotic characteristics incorporates the invasiveness of the introduced species as well as the characteristics of the recipient ecosystem and the interactions between them (Catford et al., 2009). This framework can be applied for the assessment of species on the move whereby traits of successful and damaging range-shifters could be identified along with the characteristics that might make an ecosystem particularly vulnerable to new species. Once these characteristics are more clearly defined there would be potential for identifying problematic range-shifters based on their risk posed which could in turn support further research, such as by targeting species most likely to have the greatest ecological impacts (Urban et al., 2016).

One approach to evaluating the impact of invasive species on recipient environments is through the use of an impact categorisation and scoring system. Nentwig et al. (2010) developed a Generic Impact-Scoring System (GISS) for invasive mammals in Europe as a method to compare both the environmental and economic impacts of species by delineating specific impact categories and scoring each from a minimal impact score of zero to a maximum impact score of five. The GISS method has subsequently been developed for other taxa (Kumschick & Nentwig, 2010; Vaes-Petignat & Nentwig, 2014; Van der Veer & Nentwig, 2014;) as well as advancements to incorporate stakeholder opinion (Kumschick et al., 2012) and a correlation with species traits (Evans et al., 2016). Based on the GISS concept and the widely accepted Red List of Threatened Species, Blackburn et al. (2014) developed the Environmental Impact Classification of Alien Taxa (EICAT) which is adopted by the International Union for Conservation of Nature (IUCN) to identify the magnitude of impact of invasive species. Species are classified from Minimal Concern (MC) to Massive (MV) according to semi-quantitative scenarios under specific identified impact mechanisms. These mechanisms include the chemical, structural and physical impact on ecosystems as well as species interactions such as predation, competition, and parasitism (Van der Colff et al., 2020). It has been proposed that impacts by range shifting species can occur through comparable mechanisms (Wallingford et al., 2020).

With introduced species, their success is often as a result of benefiting from novel interactions such as naïve prey and enemy release, whereas species moving as a result of climate change may have less potential for encountering novel interactions as they may shift from adjacent locations into those with a similar composition or shared evolutionary history (Wallingford et al., 2020). Nevertheless, invasive frameworks offer an interesting basis from which to investigate potential processes by which species shifting due to climate change might impact ecosystems through biotic interactions, even if these interactions are not completely novel. Furthermore, if a particular species is better able to track their ecological niche than their competitors, they may still be able to benefit from mechanisms such as enemy release (Gilman et al., 2010; Wallingford et al., 2020). Likewise, if a species on the move due to climate change shifts a large distance, particularly when traversing a biogeographic boundary that has previously limited dispersal, then there is still potential to encounter and benefit from naïve prey species (Wallingford et al., 2020). Therefore, implementing systems from invasion biology to determine impacts of species on the move may require an emphasis not only on the characteristics of the shifting species and recipient ecosystems and the mechanisms by which impacts occur but also the level of similarity between the composition of the donor and recipient environments.

Urban (2020) expressed concern that the application of invasive perspectives for range-shifting species may threaten conservation efforts and harmfully impact ecosystems, arguing that associating range-shifting species with the language and culture of eradication within invasion biology is problematic. They emphasise the value of species movements due to climate change for species persistence and the benefit for maintaining biodiversity and ecosystems more widely and the trouble of applying frameworks that are primarily concerned with deleterious impacts. However, recent developments within invasion biology have led to the acknowledgment of the potential for positive impacts of non-native species (Strubbe et al., 2011; Kumschick et al., 2012) and the development of the EICAT+ framework to categorise such positive effects, analogous to EICAT which focusses on negative impacts. These systems can be employed simultaneously to offer a value-free measurement of the impacts of invasive species or those shifting due to climate change, as either positive or negative depending on whether such changes induce an increase or decrease in a specific measurable attribute (Vimercati et al., 2022). For example, a species which causes a decrease in the population of a native taxa would be categorised as having a Moderate (MO) impact under EICAT, whilst those inducing an increase in population size would be assigned the impact category Moderate (MO+) positive impact under EICAT+.

As well as the ecological or environmental impacts of invasive species, there are also frameworks to assess the socio-economic impacts, although these are less commonly applied than their environmental counterparts. The Socio-Economic Impact Classification of Alien Taxa (SEICAT) was designed to run alongside EICAT to categorise impacts on human well-being from Minimal Concern (MC) to Massive (MV; Bacher et al., 2017). Impacts within SEICAT are considered as those which influence people's capabilities and therefore affect different constituents of human well-being (Bacher et al., 2017). These constituents are designed to be analogous to the impact mechanisms of EICAT and are divided into four categories – safety; material and immaterial assets; health; and social, spiritual, and cultural relations. As with invasive species, range-shifting species have the potential for socio-economic impacts within these categories. For example, if a crop pest shifts into a new area this could affect material and immaterial assets due to the time and money spent to compensate or combat crop losses, or a stinging animal such as a wasp or mosquito shifting into a new location has the potential to effect human health and safety (Bacher et al., 2017). Therefore, assessing the socio-economic impacts of species on the move could follow SEICAT relatively simply through a review of potential impacts within these categories and assigning relevant SEICAT scores. A potential complication, however, could be that impacts of range-shifting species might be less well-reported within literature than well-established invasive species and so it may prove difficult to find evidence for the impacts of range-shifting species in the areas in which they have arrived or examples of impacts of these shifting-species in other comparable areas (Sorte et al., 2010).

2.4. Research gaps to address

2.4.1. Integrating diverse data types for tracking species on the move

A wide range of biological recording exists including formal, structured monitoring schemes, opportunistic, unstructured records collated by specific schemes and societies, atlases, as well as wider participation in recording through the implementation of citizen science based mobile applications and websites. Therefore, species occurrence records are available through a variety of sources with different participants and information content. Detecting species on the move from such records proves difficult due to issues of their inherent biases, delays in validation, and the numerous steps involved to flag new species in new areas and then link these distribution changes to climate change.

Alternatives to traditional biological monitoring are becoming apparent, including citizen science, iEcology, and the implementation of online data for ecological research. In particular, the potential applicability of social media data has been identified to address current information needs (Pettorelli et al., 2019), with the large quantities of instantaneously available and accessible data allowing for rapid assessment of species distributions (Jarić et al., 2021). Although emerging research employing such digital data has been successful, further investigation is required to determine the utility of social media information as well as the complementarity with more traditional forms of biological recording and other citizen-science data. In particular, information is lacking as to how the distribution of species occurrence observations differ between data sources and whether different types of biological records capture different aspects of species' environmental niche.

2.4.2. Directions and dimensions of species on the move

Recommendations have been made for redistribution studies into observed and projected climate-driven distribution changes to consider directionality in climate and range dynamics (Gillings et al., 2015). Some research has surveyed species shifts across a range of axes and have reported multidirectional shifts (Gillings et al., 2015; Huang et al., 2017), however, investigations of this nature remain lacking. In addition to movement away from a univariate temperature-based approach, further research is also required to determine the influence of different climate variables and their metrics on resulting species shifts (Garcia et al., 2014). This includes investigation into the relative importance of climate averages and extremes (Garcia et al., 2014; Huang et al., 2017).

Relating unidirectional shifts towards the poles, higher altitudes, and greater depths in the oceans to climate change is easily achieved due to the temperature gradients which occur along such directions (Rubenstein et al., 2023), whereas attributing species shifts across multiple directions to a multitude of different climatic variables proves more difficult (Gillings et al., 2015). Methodologies have developed to compare environmental niches, such as between different competing species, or to analyse differences between invasive species' niche within their native and invaded range (Guisan et al., 2014). Such methods provide useful applications for comparing species niche over time during periods of environmental change, to determine if species geographic shifts due to climate change enable them to maintain a consistent climate niche over time, offering a process to attribute geographic shifts to climate change that can incorporate a range of different climate indices (Ralston et al., 2017; Sillero et al., 2022). Moreover, simultaneous quantification of species geographic distribution change alongside temporal niche comparisons offers an underutilised means of investigating species climate change response; such examinations are prudent to determine whether species are tracking climate change, whether they can keep pace to remain within historic climatic conditions, or if in fact species can instead shift their climate niche as an alternative means to survive rapid climatic changes.

2.4.3. Lack of a framework for assessing impacts of species on the move

Understanding range shifts and potential impacts is essential for informing future conservation strategies for preserving biodiversity and ecosystem functioning with a changing climate. Although parallels between invasive and climate-tracking species have been recognised within literature (Pettorelli et al., 2019; Wallingford et al., 2020), the practical application of such insights from invasion biology for species redistribution remains under-investigated. Furthermore, there are no examples of invasion frameworks having been directly employed for climate-tracking species, despite a concern of potentially severe and wide-reading anticipated impacts of species redistribution for ecosystems (Twiname et al., 2020; Wallingford et al., 2020)

Managing species on the move will rely on an intricate balancing of conservation and risk, whereby potential impacts are fully investigated. Structures need to be put in place to rapidly detect species movements due to climate change as well as evaluate potential consequences so that we are able to facilitate advantageous movements or those with less potential for negative impacts whilst inhibiting species movements most likely to adversely impact recipient environments (Wallingford et al., 2020). A framework is required to achieve this that both builds on and diverges from currently implemented national and international strategies within invasion biology and conservation more generally.

3. Research Objectives

The overarching objective of this thesis is to provide key advancements in the detection and assessment of species shifts under climate change. This is achieved through an investigation of three research areas that address the knowledge gaps:

The first is an assessment of different types of biological records. The key aim for **chapter 5** is *'to assess the level of complementarity and divergence between distribution and habitat suitability maps derived from traditional recording schemes, citizen science initiatives, and social media information'*. The hypotheses are that habitat suitability maps derived from social media records and citizen science initiatives will significantly differ from habitat suitability maps derived from traditional records (**chapter 5 H1**), and occurrences derived from social media platforms and citizen science initiatives will be more common in urban settings compared with traditional biological recording (**chapter 5 H2**).

Chapter 6 provides a novel simultaneous investigation of species geographic shifts and niche shifts and examines the alternative responses by species to either move with or adapt to changing climates. The overarching aim of this research chapter is *'to compare the response of species to climate change and determine whether species exhibit niche shifts during recent rapid changes in climatic conditions'*. It is hypothesised that not all Odonates (dragonflies and damselflies) will exhibit niche conservatism over time (**chapter 6 H1**), there will be key differences between dragonflies and damselflies in response to climate change (**chapter 6 H2**), and there will be a positive correlation between niche flexibility (higher level of change in climatic niche over time) and increase in range size (**chapter 6 H3**).

Chapter 7 provides an investigation into the potential impacts of species on the move, with the aim to '*employ the invasive EICAT and EICAT+ frameworks for climate-tracking species alongside future predictions of changing habitat suitability, providing a spatially explicit impact assessment'.* It is hypothesised that Hymenoptera will be a more problematic group, with a greater severity of negative impacts, occurring through the EICAT mechanisms reported for invasive Hymenoptera worldwide on the Global Invasive Species Database (GISD; http://www.issg.org/database) – i.e. competition, predation, transmission of disease, parasitism, and poisoning/ toxicity (**chapter 7 H1**). Not being invasive, Odonata are expected to have only minor deleterious impacts under EICAT if any, with potential mechanisms being predation, as well as competition with other Odonates (**chapter 7 H2**). Finally, it is hypothesised that species of both groups will have positive impacts through the provision of trophic resources to other native taxa, with dispersal facilitation also predicted to be relevant for Hymenoptera as many species are important pollinators (**chapter 7 H3**).

4. Methodology

4.1. Context

4.1.1. Study location

The UK makes for an excellent case study to investigate species on the move with climate change due to the vast availability of species distribution data. The UK has a historical legacy of biological recording (Sutherland et al., 2015) and is arguably the most intensively recorded country on earth (Powney & Isaac, 2015), with a vast record density (https://www.gbif.org/the-gbif-network). These biological records are predominantly collected by volunteer recorders and naturalists as well as a wide public engagement with citizen science (Pocock et al. 2015), which is increasing in support and publicity within the UK (Pettorelli et al., 2019).

In terms of investigating directionality in species distribution shifts with climate changes, Britain specifically makes for an optimal study location being a discrete geographical unit (Gillings et al., 2014). Investigation of directional changes in the UK is also interesting as climate changes are complex. In additional to typical warming trends over a latitudinal axis, warming is also typically greater in the east, and the west is broadly wetter therefore resulting distributional shifts are likely to deviate from simple poleward shifts (Gillings et al., 2014; figure 4.1).

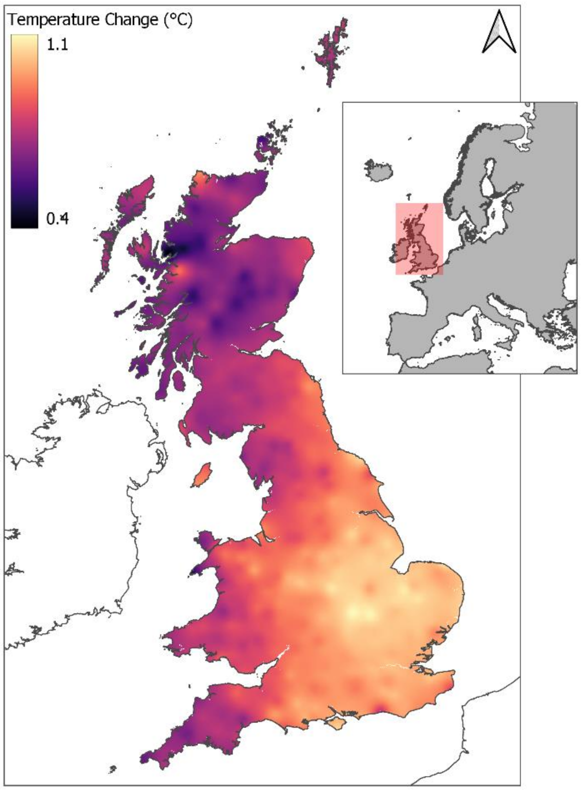


Figure 4.1. Mean temperature increase in Great Britain between 1961-1990 and 1991-2020 based on Had UK Gridded Climate Observations for the UK (Met Office, 2022).

Another reason to focus on the UK is that many beneficial frameworks are already in place that can inform this research. For example, there is an effective invasive species monitoring program with the necessary tools to enable detection of species at new locations (Pettorelli et al., 2019). Moreover, assessment tools to determine the environmental and socio-economic impacts of invasive species could be adjusted to assess potential impacts of species on the move on a species-by-species basis (Wallingford et al., 2020).

4.1.2. Study species

This thesis begins with an investigation into alternative sources of biological records utilising a case study of a single species (**chapter 5**) – the Banded Demoiselle (*Calopteryx splendens*). Then follows a comparison between species' geographic distribution shifts and climate niche changes for Britain's Odonata (dragonflies and damselflies) species (**chapter 6**). Finally, environmental impacts of species on the move are investigated for both Odonata and Hymenoptera (wasps, bees, ants, and sawflies) species (**chapter 7**).

Odonata is perceived as a charismatic taxon with a vast availability of species distribution data and well-documented ecologies (Stoks & Cordoba-Aguilar, 2012). Odonata have been recognised as a potential model taxon for investigating responses to climate change (Hassall, 2015), being particularly sensitive to environmental change and frequently employed as biological indicators for aquatic, and increasingly terrestrial environments (Miguel et al., 2017). They have a complex life cycle, metamorphosing from their aquatic, larval stage to their terrestrial adult counterparts (Stoks & Cordoba-Aguilar, 2012); they require a range of resource-rich habitats and are greatly influenced by changes in both temperature and precipitation (Hassall & Thompson, 2008; Olsen et al., 2022).

Temperature is a major determinant of Odonata distributions (Hassall & Thompson, 2008) and research indicates several European species ranges have shifted or expanded Northwards with recent climate warming (Tang & Visconti, 2020; Olsen et al., 2022; Pélissié et al., 2022). Species such as the Banded Demoiselle (*C. splendens*), Red-eyed Damselfly (*Erythromma najas*), and the Southern Emerald Damselfly (*Lestes barbarus*) have been found to be redistributing in the UK (Pettorelli et al., 2019), as well as the arrival of new species such as the Willow Emerald Damselfly (*Chalcolestes viridis*) and the Small Red-eyed Damselfly (*Erythromma viridulum*; Cranston et al., 2023). Odonata are sensitive to climatic variability and extreme weather events. Periods of drought and resulting water scarcity can lead to local extirpation of species, for example, the Azure Hawker (*Aeshna caerulea*) has previously been found to be reducing its range in south-west Scotland as the shallow pools it inhabits have become increasingly dry since 2000 (Taylor et al., 2021). Flooding events can cause mortality during emergence or result in species range expansion from the downstream transportation of larvae and eggs (Taylor et al., 2021). Warmer, drier weather conditions promote Odonatan survival during the flight season, increasing populations and supporting range expansion. As such, it is crucial for the investigation of Odonatan responses to climate change to consider a range of temperature and precipitation indices, based on both average and extreme values.

Odonates have high dispersal capabilities compared to other freshwater invertebrates and can respond rapidly to current environmental conditions (Cancellario et al., 2022; Olsen et al., 2022). Of the two suborders, Anisoptera (dragonflies) being generally larger, faster, and more robust are typically superior dispersers compared to smaller, slimmer-bodied Zygoptera (damselflies; Cancellario et al., 2022; Olsen et al., 2022). Furthermore, Anisoptera have adaptations for thermoregulation and are able to tolerate a greater temperature range compared to thermoconforming Zygoptera (Olsen et al., 2022). This interesting variability within the Odonatan order makes for an informative exploration into the potential implication of climate change. Additionally, some Odonates use lentic (standing water) and others lotic (flowing water) habitats, presenting different niche requirements. Many lentic species have had the necessity to adapt greater dispersal abilities than lotic species whose habitat is by nature more connected (Hof et al., 2006; Prescott & Eason, 2018; Cancellario et al., 2022).

Of the Odonatan order, the Banded Demoiselle (*C. splendens*) was chosen as a study species for **chapter 5** being a highly recognizable damselfly that is currently shifting its distribution in the UK due to climate change (Brooks et al., 2007; Mill et al., 2010; Cham et al., 2014; Pettorelli et al., 2019). This damselfly has a unique appearance and ease of species identification, being one of only two damselfly species in the country with coloured wings, alongside the Beautiful Demoiselle (*Calopteryx virgo*; Svensson et al., 2004). The male adult in particular has a metallic, blue body and wings with a large, melanised spot or band and therefore stark appearance (Upton et al., 2016; figure 4.2a). The female adult has a metallic green body with translucent pale green wings (figure 4.2b). Thus, this species is an ideal candidate for investigation into the use of social media and citizen science occurrence records. The Banded Demoiselle (*C. splendens*) is distinguishable from the Beautiful Demoiselle (*C. virgo*) by its darker wing colouration. The Beautiful Demoiselle (*C. virgo*) males have nearly fully melanised wings and the females have brown-tinged wings (Drury et al., 2019).

Figure 4.2. Images of the Banded Demoiselle (Calopteryx splendens) male (a) and female (b) adult damselflies. Figure 4.2a is taken by Burkard Meyendriesch (2021) and figure 4.2b by Ian Lindsay (2021). Both images were retrieved from pixabay.com and are free for use under the Pixabay Content License.

The Banded Demoiselle (*C. splendens*) is one of a few British riverine Odonates, requiring an adequate unidirectional flow for larval respiration, therefore restricted primarily to slow-flowing streams and rivers in lowland areas of southern Britain, although shifting further northward in recent years (Ward & Mill, 2005). Another important habitat requirement is bankside vegetation to allow breeding and oviposition to take place as well as providing perching sites for territorial males (Lowdon, 2015).

In assessing the potential impact of species on the move on recipient ecosystems (**chapter 7**), Hymenoptera species were studied alongside Odonata. Hymenoptera are likewise sensitive to changing climates, particularly in the case of bees whose biology depends directly on the availability of flowering plants (Ali et al., 2023). Increased temperatures can threaten bees by decreasing their foraging time and increasing parasitism through increased survival and reproduction of Varroa mites (Ali et al., 2023). Prolonged heavy rainfall events also have significant impacts for bees' foraging activity and flooding can destroy nests and hives of both wild and managed colonies (Ali et al., 2023). Temperature also impacts foraging activity for ants and wasps; however, they are able to mitigate against climate changes to an extent through nest site selection and micro-habitat utilisation (Parr & Bishop, 2022; Stabentheiner et al., 2022), and for ants through movement of the brood within the nest (Parr & Bishop, 2022). Reduced precipitation and increased temperature, however, increases desiccation risk for ants (Parr & Bishop, 2022).

The biotic interactions of the Hymenopteran order make for an interesting study of the ecological impacts of species shifts, whilst also meaning species are more susceptible to the pressure of climate change. Several Hymenopteran species are recognised for their important role in pollination in both natural and managed systems (Rasplus et al., 2010; Queffelec, et al., 2021). Subtle changes in climate can cause bee emergence and flower blooming to become desynchronised, impacting bee colonies as well as both wild and cultivated crops (Ali et al., 2023). As well as changing phenology, the range shifts induced by climate change of plants and their pollinators can occur at different rates disrupting plant-pollinator mutualisms (Ali et al., 2023). Climate change also impacts ant-plant interactions, for example with reduced seed dispersal by ants with increasing aridity (Parr & Bishop, 2022).

Hymenoptera are a highly diverse order (Queffelec et al., 2021), arguably one of the most important groups of insects both ecologically and economically (Rasplus et al., 2010) whilst also consisting of numerous invasive species (Queffelec et al., 2021). Parasitic wasps have significant roles as biological pest control agents (Rasplus et al., 2010) often with a high degree of host specificity (Queffelec et al., 2021). Hymenoptera are thus often introduced as natural enemies, as well as for pollination, with risk to recipient communities requiring further investigation (Rasplus et al., 2010). Of the Hymenoptera, ants are particularly invasive, causing substantial economic and ecological damages (Rasplus et al., 2010), such as the Invasive Garden Ant (*Lasius neglectus*). Another notable alien species is the Asian hornet (*Vespa velutina*) which is invasive in Europe and is a veracious predator or pollinating insects such as hoverflies and bumblebees (Keeling et al., 2017); this species been confirmed in Great Britain and is of particular threat to the European honeybee (*Apis mellifera*). With the prevalence of invasive species within the Hymenopteran order, species on the move might be similarly problematic for recipient ecosystems.

4.2. Datasets utilised

4.2.1. Species occurrence data

(a) Banded Demoiselle occurrences

Chapter 5 provides a review of difference sources of biological records for the Banded Demoiselle (*C. splendens*). Recent records were sourced from online portals and separated by type as well as gathered from social media platforms (section 4.2.3 table 4.1).

Traditional and citizen science

Species occurrence records were downloaded from both the Global Biodiversity Information Facility (GBIF, 2021) and the National Biodiversity Network (NBN) Atlas (British Dragonfly Society Recording Scheme, 2021; National Biodiversity Network Trust, 2021) to incorporate the broadest range of verified data available. The UK NBN collates observations from a variety of sources and feeds directly into GBIF, however, it is typically updated more frequently than GBIF while some data providers or individuals might share records directly with GBIF rather than through the NBN Atlas (Callcutt et al., 2018). Records were selected from 2010 onwards for comparison with social media datasets that do not have such an extensive history of biological recording and providing approximately a decade of data for comparison. Biological records from the British Dragonfly Society (BDS) Recording Scheme (excluding records from iRecord), Local Environmental Record Centres (LERC) as well as various national and international trusts and organisations were labelled as 'traditional'. These records are typically collated and verified by experts and Country Recorders, ensuring data accuracy and reliability. Records from both the iRecord and iNaturalist platforms were labelled as 'citizen science', as they can be submitted by anyone, regardless of their level of expertise.

Social media

Records were collected from social media platforms (Facebook, Twitter, and Flickr) using the search terms "Banded Demoiselle" as well as "*Calopteryx splendens*" (Appendix A). For Twitter (table A.1) and Facebook (table A.2) this involved a manual search (completed between 13/01/2022 and 04/04/2022, for approximately 1.5 hours a day), with biological records consisting of an identifiable photograph or video. These records included either a tagged location or a mention of location within the content of the post, as well as a date for the observation if provided (otherwise the date the content was shared). Latitude and longitude information was generally preferable, allowing for precise placement of species occurrences. However, this information was not available for Twitter or Facebook records. Around 23% of the Twitter records found included a tagged location label, however, this was typically a city or town level. As such, records from Twitter and Facebook were manually checked and geo-referenced by determining all the 1-km British National Grid squares that covered the spatial extent of the location description provided by the user. Although more imprecise than tagged geo-locations, this ensured that the location information included was where the observation occurred (as opposed to where the photograph was uploaded). Searches yielded 95 results from Twitter and 331 from Facebook, which covered 295 and 867 1-km grid squares, respectively. These 1-km grid squares were included as Banded Demoiselle (*C. splendens*) occurrences in subsequent Species Distribution Models (SDMs). For each social media occurrence, spatial precision (estimated to the nearest square km) was recorded in the final dataset.

For Flickr (table A.3), records were collated with the Flickr Application Programming Interface (API) using the Flickr.photos.search (2021). Initial searches yielded 1,316 results with location information as well as date recorded and posted that were extracted in R using the package 'FlickrAPI' (Ando & Pousson, 2022). These results were manually verified, with 1,223 observations remaining once records observed outside of the relevant time frame or study location as well as irrelevant or misidentified species observations were removed.

(b) Odonata and Hymenoptera occurrences

For **chapters 6** and **7**, species occurrence records for Odonata species in Great Britain between 1961 and 2020 were downloaded from both the GBIF (2023) and the NBN Atlas (British Dragonfly Society Recording Scheme, 2023; NBN Trust, 2023). These observations are primarily opportunistic, unstructured records but incorporate a range of information sources including both traditional and citizen science records. This data enabled calculation of trends for Odonata species throughout this period. Records were also separated into two distinct periods T1 (1961-1980) and T2 (2001-2020), enabling a temporal comparison of species' geographic and niche dynamics. The most recent period, T2 (2001-2020), also informs SDMs in **chapter 7** and therefore is referred to interchangeably as 'current occurrence'.

For implementation in **chapter 7**, Hymenoptera occurrences were also acquired for the two periods (T1 and T2) from both the GBIF (GBIF, 2023) and the NBN Atlas (Biological Records Centre, 2023; Bumblebee Conservation Trust, 2023; BeeWalk bumblebee distributions for Great Britain 2008-2022, 2023; Bee, wasp and ant (Hymenoptera: Aculeata) records verified via iRecord, 2023; NBN Trust, 2023; Sawfly (Symphyta) records from iRecord, 2023), as well as additional data for Hymenoptera acquired directly from the Bees, Wasps and Ants Recording Society. As with the Odonata records, the Hymenoptera data originated from a variety of sources, different organisations, and initiatives.

4.2.2. Spatial data

(a) Climate data

Species Distribution Modelling (SDM): current and projected climate

Under the assumption that a species' range responds to the long-term averages of climate conditions (Taheri et al., 2020), monthly minimum and maximum temperature as well as monthly precipitation data for the period 1991 to 2020 was accessed from the Met Office at a 1-km resolution for **chapter 5** (Met Office, 2022) and a 5-km resolution for **chapter 7** to be consistent with climate projections (Met Office et al., 2023). These datasets are required to generate a series of 19 bioclimate variables which were calculated using the biovars function in the R package 'dismo' (Hijmans et al., 2021). These bioclimatic variables are known to be biologically meaningful and informative for Odonatan and Hymenopteran distributions (Collins et al., 2017; Abbott et al., 2022), as they capture information on annual trends, seasonality, and limiting environmental factors and are widely employed for SDM (Manzoor et al., 2018).

For **chapter 7**, climate projections from the United Kingdom Climate Projection 2018 (UKCP18) project were also downloaded at a 5-km resolution from the Met Office Hadley Centre (2019) for the period 2020-2040 under a high emissions scenario – Representative Concentration Pathway 8.5 (RCP8.5) to provide an upper bound for potential species impacts. Total rainfall, minimum temperature, and maximum temperature variable projections were retrieved from each of the twelve Met Office Hadley Centre models, and a mean calculated for each variable across these models. As with the current climate data, these monthly variables were subsequently employed to generate the 19 bioclimate variables in R (table 4.1).

Climate niche analyses: climate averages and extremes

To compare species realised climatic niche between T1 and T2 (**chapters 6** and **7**), bioclimate data for these periods was employed to capture changes in climate averages. Despite their potential importance in shaping species distributions (Huang et al., 2017), climate extremes have received little attention when assessing the effects of climate change on biodiversity (Garcia et al., 2014). To address this, six indices that capture the intensity and frequency of climate extremes were also considered for the niche analyses, based on the work from the Expert Team on Climate Change Detection and Indices (ETCCDI). These indices were calculated from daily temperature and precipitation data accessed from the Met Office (Hollis et al., 2019), and included: the maximum of daily maximum temperature (TXx); the minimum of daily minimum temperature (TNn); the number of frost days (FD; daily minimum temperature below 0°C); the number of summer days (SU; daily maximum temperature above 25°C); the number of consecutive dry days (CDD; precipitation below 1 mm); and the maximum 5-day precipitation (Rx5day; maximum precipitation in five consecutive days). As with the bioclimate variables, each index was calculated on an annual basis and then averaged for each period. Calculations were completed using R package 'ClimInd' (Reig-Gracia et al., 2021).

(b) Environmental data

Predictor variables for the Banded Demoiselle

In addition to climate, the set of environmental variables considered to shape the distribution of Banded Demoiselle (*C. splendens*) in the UK included topography, landcover type, vegetation productivity, and level of urbanisation (table 4.1). Slope was extracted from the Ordnance Survey (OS) Terrain 50 Digital Terrain Model (DTM) accessed from EDINA Digimap (OS Terrain 50, 2013); slope is important for Odonata species due to its influence on water velocity, O2 content, weathering, channel substrate size, and organic matter composition (Collins & McIntyre, 2015) and of particular importance to the Banded Demoiselle (*C. splendens*) that favours slow flowing rivers.

To capture the aquatic element of the Banded Demoiselle's (*C. splendens*) niche the percentage cover at 1-km resolution of the freshwater aggregate class was extracted from the Centre for Ecology and Hydrology (CEH) 2015 Land Cover Map accessed from EDINA Digimap (Land Cover

Map 2015, 2017). A Water and Wetness Probability Index (WWPI) product coordinated by European Environment Agency (EEA) Copernicus programme was also acquired which indicates the occurrence of water and wet areas as a continuous probability at 20-m resolution based on observations between 2009 and 2015 (Langanke et al., 2018).

Normalised Difference Vegetation Index (NDVI; Pettorelli, 2013) Long Term Statistics (LTS) version 2.2. were also included from the Copernicus Global Land Service (CGLS) at a 1-km resolution (Toté et al., 2021). These statistics include the minimum, median, maximum, average, and standard deviation calculated from 10-daily NDVI values throughout 1999 to 2017 derived from Spot-4, Spot-5 and Proba-V satellite imagery. In addition, the CGLS 100-m resolution tree cover density for the reference year 2012 was included (European Environment Agency, 2018). These variables should account for the significance of vegetation on the Banded Demoiselle (*C. splendens*) distribution, where vegetation influences territory selection and eggs are laid into aquatic emergent vegetation (Ward & Mill, 2005). Therefore, vegetation cover indicated by level of "greenness" and tree density is likely to be informative of Banded Demoiselle (*C. splendens*) habitat suitability.

Finally, as an indication of urbanisation, annual composites of visible night light version 2 were acquired for the years 2014 to 2018 from the Earth Observation Group (Elvidge et al., 2021) and averaged across these years.

Forest cover for the gall wasps (Hymenoptera)

For **chapter 7**, information on the distribution of broadleaf forests was also obtained as three identified climate-tracking Hymenoptera species in are gall wasps whose distribution explicitly relies on the presence of oak woodland; however, data was not available at the species level. Therefore, the National Forest Inventory GB 2020 was downloaded from the Forest Commission Open Data (2023) which is a shapefile file of Great Britain's woodland. To prepare this data on the same resolution as the climatic data, the broadleaf area was first extracted, and the percentage coverage calculated for each 5-km British National Grids using QGIS. Although incorporating other non-oak woodland areas, such as ash and beech, may not be ideal, broadleaf woodland information could still provide valuable insight into gall species' potential habitat suitability. Therefore, it might be beneficial to retain this information if deemed important when reviewing the environmental variables.

Altitude and landcover

Altitude was derived from the OS Terrain 50 digital terrain model available on the OS OpenData Downloads platform (https://osdatahub.os.uk/downloads/open). This was obtained to enable comparison between species distribution in terms of mean altitude between periods. To further investigate species habitat, 10 aggregate classes of the CEH 2015 land cover map was accessed from EDINA Digimap (Land Cover Map 2015, 2017).

Recorder effort proxies

To account for the spatial bias of recorder effort during modelling (**chapter 7**), several potential sampling effort proxies were acquired – percentage cover of built-up areas and gardens, distance to the nearest road, population density, and visible night light (table 4.1). Assuming that measurements relating human presence and accessibility are correlated with the level biological recording, these variables can be employed to account for bias in SDM. To achieve this SDMs are built using proxy data, then proxy variables are adjusted to a unform level across space when projecting species habitat suitability, effectively reducing recorder bias. Percentage cover of built-up areas was extracted from the UK CEH 2021 Land Cover Map at a 1-km resolution (Marston et al., 2022). The OS Open Roads (2023) vector dataset was downloaded for the UK from EDINA Digimap, and the distance from each 5-km grid cell in the study area to the nearest road was calculated in QGIS. Residential gridded population data was also acquired from Digimap, which is based on the 2011 Census and 2015 Land Cover Map (Reis et al., 2017). Annual composites of visible night light version 2 for the years 2016 to 2020 were downloaded for the Earth Observation Group (Elvidge et al., 2021) and averaged across these years.

4.2.3. Data summary

Table 4.1. provides a summary of all the datasets employed throughout this thesis including species occurrence as well as spatial climate and environmental data. The source of each dataset is indicated as well as the use in the relevant data chapter.

Table 4.1. Summary of data employed throughout this thesis. Abbreviations are as follows: C5 – Chapter five, C6 – Chapter six, C7 – Chapter seven, Res. – Resolution, BDS - British Dragonfly Society, LERC - Local Environmental Records Centres, NT – National Trusts, SDM – Species Distribution Model, GBIF – Global Biodiversity Information Facility, NBN – National Biodiversity Network, BWARS – Bees, Wasps and Ants Recording Society, ETCCDI – Expert Team on Climate Change Detection and Indices, OS – Ordnance Survey, CEH – Centre for Ecology and Hydrology, EEA – European Environment Agency, CGLS – Copernicus Global Land Service, EOG – Earth Observation Group, NERC – Natural Environment Research Council, EIDC – Environmental Information Data Centre.

OCCURRENCE DATA				
	Dataset	**No. records**	**Source**	**Use**
Banded Demoiselle (2010-2022)	*Traditional*	6,759	BDS, LERCs NTs	SDM (C5)
	Citizen Science	9,646	iRecord, iNaturalist	
	Social Media	2,026	Facebook, Twitter, Flickr	
Odonata	*1961-2020*	695,180	GBIF, NBN Atlas	Trends (C6)
	1961-1980	33,103	GBIF, NBN Atlas	Distribution & niche change (C6 & C7)
	2001-2020	499,519	GBIF, NBN Atlas	Distribution & niche change (C6 & C7) SDM (C7)
Hymenoptera	*1961-1980*	16,802	GBIF, NBN Atlas, BWARS	Distribution & niche change (C7)
	2001-2020	593,305	GBIF, NBN Atlas, BWARS	Distribution & niche change (C7) SDM (C7)

RAW CLIMATE DATA				
Variable	**Period**	**Res.**	**Source**	**Use**
Monthly maximum air temperature *Monthly minimum air temperature* *Monthly precipitation*	1961-1980 1991-2020 2001-2020 1991-2020 2020-2040	1km 1km 1km 5km 5km	Met Office	Generate Bioclimate variables
Daily maximum air temperature *Daily minimum air temperature* *Daily precipitation*	1961-1980 2001-2020	1km 1km	Met Office	Generate ETCCDI variables
BIOCLIMATE VARIABLES				
Variable	**Period**	**Res.**	**Source**	**Use**
Annual Mean (Bio1) *Mean diurnal range (Bio2)* *Isothermality (Bio3)* *Temp. Seasonality (Bio4)* *Max. temp. of warmest month (Bio5)* *Min. temp. of coldest month (Bio6)* *Temp. annual range (Bio7)* *Mean temp. of wettest quarter (Bio8)* *Mean temp. of driest quarter (Bio9)* *Mean temp. of warmest quarter (Bio10)* *Mean temp. of coldest quarter (Bio11)* *Annual precipitation (Bio12)* *Precipitation of wettest month (Bio13)* *Precipitation of driest month (Bio14)* *Precipitation seasonality (Bio15)* *Precipitation of wettest quarter (Bio16)* *Precipitation of driest quarter (Bio17)* *Precipitation of warmest quarter (Bio18)* *Precipitation of coldest quarter (Bio19)*	1961-1980 2001-2020	1km 1km	Monthly climate data	Niche comparison (C6, C7)
	1991-2020	1km	Monthly climate data	SDM (C5)
	1991-2020 2020-2040	5km 5km	Monthly climate data	SDM (C7)

ETCCDI VARIABLES				
Variable	**Period**	**Res.**	**Source**	**Use**
Max. of daily maximum temp. (TXx) *Min. of daily minimum temp. (TNn)* *No. of frost days below 0°C (FD)* *No. of summer days above 25°C (SU)* *No. of consecutive dry days (CDD)* *Max. precipitation in five consecutive days (RX5day)*	1961-1980 2001-2020	1km 1km	Daily climate data	Niche comparison (C6, C7)
ENVIRONMENTAL VARIABLES				
Variable	**Period**	**Res.**	**Source**	**Use**
Slope	2014	50m	OS Terrain 50	SDM (C5)
Percentage freshwater cover	2015	1km	CEH	SDM (C5)
Water and Wetness Probability Index (WWPI)	2009-2015	20m	EEA	SDM (C5)
Normalized Difference Vegetation Index (NDVI)	1999-2017	1km	CGLS	SDM (C5)
Tree cover density	2012	100m	CGLS	SDM (C5)
Visible night light	2014-2018	15 sec	EOG	SDM (C5)
Percentage broadleaf forest cover	2020	5km	Forest Commission	SDM (C7)
Altitude	2023	50m	OS Terrain 50	Distribution Change (C6)
Landcover	2015	1km	CEH	Habitat (C5)
RECORDER PROXIES				
Variable	**Period**	**Res.**	**Source**	**Use**
Percentage cover of built-up areas	2021	1km	CEH	SDM (C7)

Distance to nearest road	2023	5km	OS Open Roads	SDM (C7)
Residential population	2017	1km	NERC EIDC	SDM (C7)
Visible night light	2016-2020	15 sec	EOG	SDM (C7)

4.3. Data preparation

4.3.1. Species occurrence data

All sets of species occurrence records were cleaned using the R package ‘CoordinateCleaner’ to flag and remove erroneous or duplicate results (Zizka et al., 2021). Potential data entry errors and failed geo-referencing were flagged by checking for equal latitude and longitude values as well as zeros in the coordinates. Coordinates matching country centroids and biodiversity institutions were also removed using reference data within the package, as these are likely incorrect records or records of captured individuals (Zizka et al., 2019).

(a) Sampling effort

Filtering

To compare species distribution between periods sampling effort must be accounted for because typically recording intensity has increased over time (Isaac & Pocock, 2015), which could be misconstrued as an increase in species’ geographic distribution size. To enable an accurate comparison between periods, occurrence data was filtered to only include localities that have been adequately sampled during both T1 and T2, by employing the Frescalo (FREquency SCAling LOcal) method (Hill, 2012), using the frescalo function within the R package ‘sparta’ (August et al., 2015). This method was also applied to determine species’ trends (see section 4.4.1.b) where it is explained in further detail, however, is introduced briefly here as a means of filtering occurrence data. The Frescalo method relates biological records of the target species to a suite of local benchmark species within the same taxonomic group that vary between neighbourhoods based on the similarity between record sites according to landcover. Using this method, the number of species expected to be recorded in each British National Grid hectad (10 km^2) can be estimated, and then by comparing the number of observed species for each period to the value predicted after accounting for variation in recording effort, a subset of adequately sampled hectads can be identified. These adequately sampled hectads are defined as those which reported at least 25% expected species at both T1 and T2, following previous research (Hordley et al., 2023). Therefore, subsequent distribution analyses for each species only incorporate those localities that have been surveyed to an appropriate level at both times and exclude localities which are only adequately sampled more recently, for example.

Bias covariate correction

Sampling bias of opportunistically sampled data can severely distort results of SDMs that rely on the assumption that sampling bias and probability of detection are approximately even over a given area (Bird et at., 2014; Johnson et al., 2021). To account for this sampling bias when applying modelling algorithms, a 'bias covariate correction' method was employed (Warton et al., 2013; Chauvier et al., 2021), where proxies for sampling effect are used to correct for bias. Several potential sampling effort predictors were reviewed (see section 4.2.2.b above) and the most important variables retained (see section 4.5.1.b below), resulting in the inclusion of the distance to nearest road and visible night light as the chosen recorder effort proxies (**chapter 7**). Model algorithms were trained with these variables alongside selected predictor variables, then when projecting species current and future habitat suitability these recorder proxies were set to the median value across the entire study area, following the protocol of Warton et al. (2013). Thereby recorder effort is included when building SDMs and adjusted to a consistent value throughout the study area during SDM projections, thus accounting for the spatial bias of occurrence data. See section 4.5 for full details of the SDM modelling process.

4.3.2. Spatial data

Spatial data was reprojected to the British National Grid projection and aggregated to a consistent resolution for implementation within SDMs, at a 1-km resolution for the Banded Demoiselle (*C. splendens*; **chapter 5**), and a 5-km resolution across Odonata and Hymenoptera (**chapter 7**) to match the highest resolution climate projections available. This was achieved using the projectRaster and aggregate functions in R package 'raster' (Hijmans & Van Etten, 2012). All predictor variables and recorder effort proxies were scaled to a mean of zero and a standard deviation of one for SDM implementation. Predictor distributions were checked for any significant skew and a log transformation applied where a strong skew was identified.

4.4. Geographic distribution change

4.4.1. Occupancy trends

(a) Telfer

For **chapter 7**, study species were selected as those which have increased in range size during recent climate change, by comparing Odonata (Appendix B, table B.1) and Hymenoptera (table B.2) species' distributions between T1 and T2. Here change in occupancy was determined by employing the method devised by Telfer et al. (2002), an index of the deviation from the change observed across all species within that taxonomic group, thus accounting for uneven sampling effort over time. The Telfer index for each species is the standardised residual from a linear regression across all species and is therefore a measure of relative change. This method is relatively simplistic, and when comparing different methods for extracting signals of occupancy change from opportunistic data has been found to be among the most robust methods (Isaac et al., 2014); other more complex and computationally taxing methods, have been found to be more powerful (Isaac et al., 2014), however, Telfer’s index was deemed sufficient here with the focus being on simply determining whether species’ range size has increased or decreased.

(b) Frescalo

Odonata species occupancy trends were determined by employing the Frescalo method introduced in section 4.3.1.a (Hill, 2012), conducted with the frescalo function in the R package ‘sparta’ (August et al., 2015). One approach to accounting for varying recorder effort is employing benchmark species, which are widespread and stable in their occurrence throughout the study area (Hill, 2012). Accordingly, the records of other species can be related to these benchmark species, however, across a large study area it is difficult to select ubiquitous species, and variation across the study area means that presence at one locality cannot be any indication of recorder effort at another. To counteract this, Hill (2012) developed the Frescalo method which employs benchmark species, however, they are not consistent across a study area and instead vary between neighbourhoods spatially and according to similarity between sites. Here similarity is based on the 2015 UK Land Cover Map data accessed through the R package. This allows for a determination of species with ought to be reported at each locality based on species presence in neighbouring localities and the level of similarity between sites, hence a measurement of recorder effort can be determined by comparing lists of species predicted to be present to those which were actually reported; this method was implemented to specify adequately sampled localities to filter occurrences for further analysis (see section 4.3.1.a).

To extract species trends over time, local benchmark species are defined as the 27% most frequent Odonatan species in a given neighbourhood, in accordance with previous research (Hill, 2012; Eichenberg et al., 2020). Time factors (TFactors) were generated to assess the reporting rate of a focal species at a given time relative to these local benchmark species. As it is advised that benchmark species should be those whose occupancy is relatively stable over time, a re-analysis was conducted, which excluded the 5% most strongly increasing or decreasing species as potential benchmark species (Hill, 2012). Changes in occupancy, defined as changes in the number of hectads where a given species is expected to be present (Hill, 2012), can be analysed by separating occurrence data into time periods. Here, two approaches were employed, first by pooling occurrences into two equal time periods, T1 (1961-1980) and T2 (2001-2020), consistent with other distribution and climate niche analyses, termed Frescalo_p following Isaac et al. (2014). For Frescalo_p, TFactors were calculated, with Z-tests applied to determine the significance of the trend. In the second approach, termed Frescalo_5y, TFactors were extracted for twelve five-year time periods throughout the duration of the entire study period (1961 to 2020 inclusive) and occupancy trends were extracted through linear modelling of TFactors. Species were identified as displaying significant trends in occupancy if either the Frescalo_5y or Frescalo_p approach yielded significant results, with the results at the higher temporal resolution of Frescalo_5y taken in the first instance. In reviewing different methods of extracting trends from opportunistically and unevenly sampled occurrence records, Isaac et al. (2014) found the Frescalo method to be among the most robust and powerful, advocating for methods such as these which directly model the data collection process.

4.4.2. Distribution centre

Species geographic shift was analysed in terms of the change in the geographic centre of species' distribution between periods T1 and T2 – a simplistic measure that nonetheless allows for a multidirectional assessment. The mean latitude and longitude for each time period were extracted using QGIS from species presence observations at 1-km British National Grid resolution, including only those occurrences within the previously determined adequately sampled hectads. The distance and direction between species centre at T1 and T2 were then determined using R package 'geosphere' with the functions distVincentyEllipsoid and bearing, respectively (Hijmans, 2022). This method gives an approximation of species distributional change throughout Great Britain and provides an indication of the distance and rate with which such changes can occur over a 40-year period that can be compared across species. Altitude was extracted for each of the species' observations using QGIS and the difference in mean altitude between periods computed for each species.

4.5. Species Distribution Modelling (SDM)

Correlative SDMs were employed to review differences between current projected habitat suitability for the Banded Demoiselle (*C. splendens*) throughout Great Britain, according to different types of occurrence records (**chapter 5**). Therefore, models were built with different sets of occurrence data with the same climate and environmental predictor variables, keeping models consistent to determine any differences in model output according to the different types of biological recording – traditional, citizen science, and social media.

SDMs were also implemented to predict future changes in Odonata and Hymenoptera study species' distributions under climate change (**chapter 7**); to achieve this species habitat suitability in Great Britain was projected for each species under current climate and according to projected future climate under a high emission warming scenario (RCP8.5). This section therefore outlines the broad SDM methodologies applied as well as presenting the specific approaches undertaken for the two research chapters. All SDMs were implemented using the R 'biomod2' package (Thuiller et al., 2021).

4.5.1. Preliminary modelling

(a) Pseudo-absence generation

All species occurrence data employed throughout this thesis is presence-only, therefore necessitating the generation of pseudo-absence occurrences for the implementation of certain SDM algorithms. For the Banded Demoiselle (*C. splendens*), five sets of 10,000 pseudo-absence points were randomly selected from the background data for each separate dataset (traditional, citizen science, and social media), a quantity approximately matching the most numerous occurrence dataset, to be broadly appropriate across SDM techniques (Barbet-Massin et al., 2012). A number of pseudo-absence sets were implemented to ensure their composition was not impacting results. To reduce the potential of selecting pseudo-absences within the same niche as presences, pseudo-absences were placed at least 1.5 km away from any observed occurrences that have a coordinate uncertainty of up to 1-km.

To model the changing distribution of Odonata and Hymenoptera species, 1,000 pseudo-absence points were randomly selected for each species. As these models were performed at a coarser resolution a lower quantity of only one set of pseudo-absence points was selected, as the broad distribution of some species within the study area limited the availability of background data, whilst 1,000 points was approximately proportional to the less numerous occurrence datasets and therefore would be broadly appropriate across all study species (Barbet-Massin et al., 2012). Pseudo-absence points were likewise generated at a distance from species presences, in this case at a distance of at least 5.5 km with the background data at a 5-km resolution.

(b) Predictor selection

The acquired climate and environmental variables which might influence the distribution of the Banded Demoiselle (*C. splendens*) or other Odonata and Hymenoptera species were reduced to only include those which are the most important as well as excluding highly correlated variables. Preliminary SDMs were implemented to determine variable importance to guide both predictor selection as well as determine the most appropriate recorder effort proxies to incorporate. For the Banded Demoiselle (*C. splendens*) preliminary models included occurrence data across all the different data types (**chapter 5**). For **chapter 7**, preliminary models consisted of projections under both current and future climates for a subset of four species, the most and least densely recorded species for both taxonomic groups – the White-legged Damselfly (*Platycnemis pennipes*), the Common Darter

(*Sympetrum striolatum*), the Blunt Tailed Digger Wasp (*Crossocerus dimidiatus*) and the European Wasp (*Vespula vulgaris*).

Variable importance was determined by computing the Pearson's correlation between SDM predictions made with a given variable and with the variable replaced with a randomised input, with variable importance averaged from five permutations. The Pearson's correlation coefficient was calculated between each pair of predictor variables and where the coefficient was greater than 0.7, only one variable was retained guided by variable importance. Including co-varying predictors above this threshold results in increased uncertainty and disagreement among projections (Dormann et al., 2012; Brun et al., 2019).

In cases where it was unclear which set of co-varying predictors should be kept, two separate models were run with each set of potential variables, and the set that contributed to a more accurate model fit was kept, assessed by True Skill Statistic (TSS); for further details on model evaluation see section 4.5.2.a. As a final check to ensure no correlated predictor variables were included, the Variable Inflation Factor (VIF), a measure of multicollinearity, was calculated for each model before computation, to ensure that VIF was less than six, which is deemed acceptable (Guisan et al., 2017).

These preliminary screening steps resulted in a final set of predictors for the Banded Demoiselle (*C. splendens*) consisting of annual mean temperature (Bio1), isothermality (Bio3), mean temperature of the wettest (Bio8) and driest quarters (Bio9), total annual precipitation (Bio12), slope, percentage freshwater cover, WWPI, mean NDVI, and percentage tree cover (**chapter 5**). The sampling effort proxy variables which were important ($1 - r > 0.05$, where r is the Pearson's correlation coefficient) and retained for the final model was distance to nearest road.

For the SDMs of the Odonata and Hymenoptera species, the predictor variables selected through this screening process were the minimum temperature of the coldest month (bio6), temperature annual range (bio7), mean temperature of the warmest quarter (bio10), and precipitation of the wettest quarter (bio16), as well as percentage broadleaf woodland cover (**chapter 7**). The recorder effort variables which had an important influence were distance to nearest road and visible night light.

(c) Algorithm selection

There was no *a priori* reason to select one family of models over another, so all were trialled through preliminary modelling and compared in terms of habitat suitability outputs and performance. Since all performed similarly, ensemble models were built with output from all high performing models, regardless of family. As such, a set of six modelling techniques were selected including three machine-learning methods, generalized boosting model (GBM), random forest (RF), and maximum entropy (MAXENT); two regression methods, generalized linear model (GLM) and multiple adaptive regression splines (MARS); and finally, a recursive partitioning method, classification tree analysis (CTA). This suite of models was applied for all species using the default 'biomod2' settings for each model algorithm.

4.5.2. Ensemble modelling

(a) Model evaluation

To review Banded Demoiselle (*C. splendens*) habitat suitability projections, five sets of validation models were created for each data type, where 20% of the species occurrences – including both presence and pseudo-absence points – were randomly selected and set aside for evaluation with the remaining 80% occurrence data retained for model training (**chapter 5**). Model performance was assessed with TSS, which provides a threshold-independent measure of accuracy (Allouche et al., 2006). TSS has several documented drawbacks (Leroy et al., 2018), notably its dependence on prevalence; however, as a balanced approach was applied here where the number of pseudo-absences was set to match the number of presences, this reduces the chance of bias when using TSS results and allows easier comparison between different models as prevalence is held constant. This the recommended approach when attempting to maximise discrimination in SDMs (Steen et al., 2020). Several alternative metrics were also calculated to provide an overall summary of performance and potential bias. These included the Area Under the Receiver Operating Characteristic (ROC) Curve (AUC); frequency bias, the ratio between observed and predicted presences; accuracy, the fraction of occurrences correct; and finally, Cohen's Kappa coefficient, a measure of model accuracy which corrects for accuracy expected to occur by chance (Allouche et al., 2006).

The same performance metrics were employed across Odonata and Hymenoptera species models (**chapter 7**). In this case, following developments of the 'biomod2' package, a five-fold cross-validation technique was applied, similarly splitting occurrences into 20% and 80% proportions for testing and training models, however, ensuring a different selection of occurrence data is set aside each time and hence all occurrences are ultimately employed to both test and train algorithms.

(b) Predicting Banded Demoiselle presence according to different datasets

For the Banded Demoiselle (*C. splendens*) ensemble models for each occurrence dataset were built combining all individual models with a TSS value greater than 0.6, considered to be useful to excellent (Komac et al., 2016), and weighing model contribution according to TSS scores. As the aim was to compare results according to different sources of biological records, the final model projections presented are based upon all available occurrence data, without any presences or pseudo-absences set aside for internal validation. This is to ensure the final parameter estimates are built with the maximum information and therefore lower uncertainty in parameter estimates and projections. It was verified that these final models were sufficiently similar to the validation models to ensure the final projections were similarly robust and that the internally validated models are representative of the final results. This was determined using a Spearman's correlation between the projected habitat suitability of the five validation models and the final models for each data source. Each ensemble model projection of habitat suitability was converted into binary presence-absence maps using thresholds that were selected to maximise the combined sensitivity and specificity scores (Liu et al., 2016).

(c) Projecting species current and future habitat suitability

Across Odonata and Hymenoptera species, ensemble models were built combining all individual models with a TSS value greater than 0.4, considered to be useful (successfully able to discriminate suitable from unsuitable areas; Zhang et al., 2015; Soultan & Safi, 2017), and weighing model contribution according to TSS scores. The current and future distribution of species was therefore projected by employing these ensemble models by training model algorithms on species' current distribution under recent climate conditions and subsequently predicting habitat suitability across Great Britain under these current climate conditions as well as under future projected climate conditions. These habitat suitability projections were converted to binary presence-absence maps by selecting a threshold maximizing the TSS score of ensemble model predictions for each species (see Appendix C, figures C.1 to C.17).

4.6. Climate niche change

Species' niche change between T1 (1961-1980) and T2 (2001-2020) was determined using an ordination approach, which has been shown to quantify niche overlap more accurately than reciprocal ecological niche models when investigated using virtual species (Guisan et al., 2014). Ordination is based on direct species observations, whereby the difference in the climatological attributes of the sites where each species was recorded in each period were compared across the available environmental space (i.e., the climatological conditions throughout the study area across both time periods), as opposed to the geographic space. The environmental space is typically a reduced multidimensional space in which a combination of initial variables is represented across two axes (Guisan et al., 2014). A summary of Odonata and Hymenoptera species climate niche change is provided in Appendix B figure B.1 and B.2, respectively.

4.6.1. Principal Component Analysis (PCA)

To reduce the dimensionality of the available climatic data, a Principal Component Analysis (PCA) was conducted for the 15 temperature and 10 precipitation variables, calibrated with data for the entire study area across both periods. The first two principal components were retained from the PCA, capturing 77.2% of the variability across all climate data (figure 4.3). PC1 accounted for 58.7% of the variance and PC2 for 18.5%. For PC1, the highest positive correlation was with precipitation of the driest month (Bio14) and quarter (Bio17) and the highest negative correlation with maximum temperature of the warmest month (Bio5) and mean temperature of warmest quarter (Bio10; table 4.2; figure 4.4a). This component can thus be understood as contrasting wet conditions from warmest ones. The second principal component integrates information relative to the coldest conditions within a year with major contributions from seven temperature variables, with strong positive correlations with minimum of daily minimum (TNn), minimum temperature of coldest month (Bio6) and mean temperature of the coldest quarter (Bio11) and a strong negative correlation with number of frost days (FD; table 4.2; figure 4.4b).

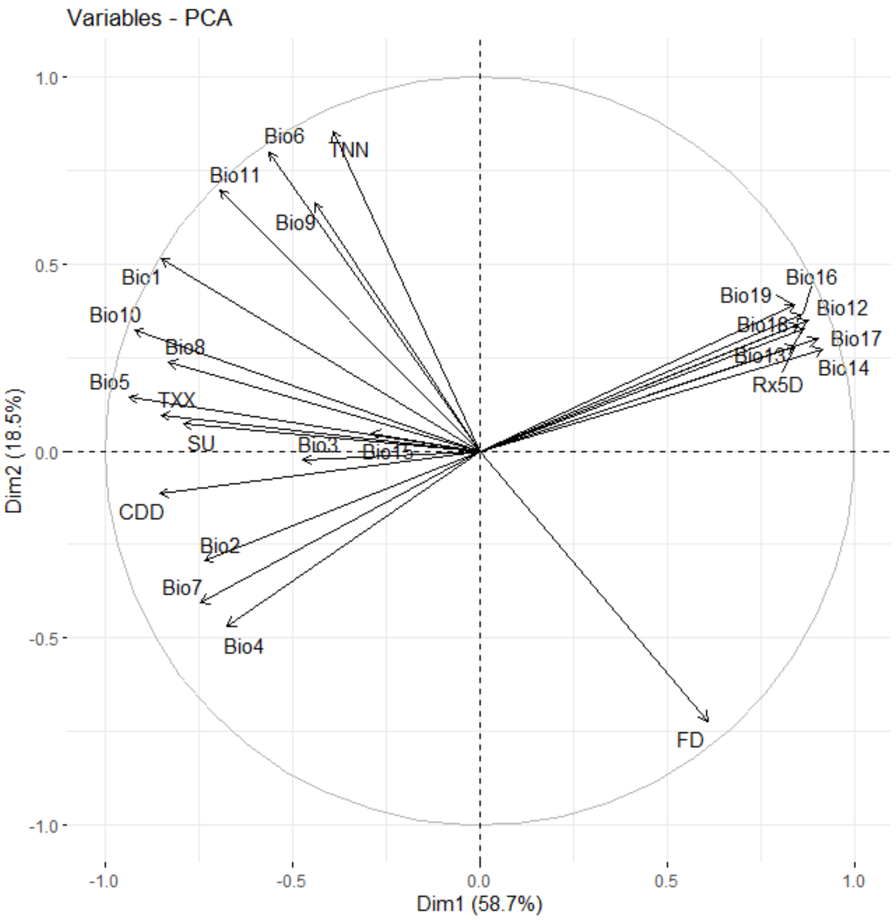


Figure 4.3. The first two principal components of the PCA of the 25 climatic variables.

Table 4.2. Description of climate variables and principal component coordinates.

Variable	Description	PC1	PC2
Bio1	Annual mean temperature	-0.852	0.514
Bio2	Mean diurnal range	-0.735	-0.294
Bio4	Temperature seasonality	-0.675	-0.470
Bio5	Maximum temperature of warmest month	-0.938	0.143
Bio6	Minimum temperature of coldest month	-0.563	0.798
Bio7	Temperature annual range	-0.746	-0.406
Bio8	Mean temperature of wettest quarter	-0.831	0.237
Bio9	Mean temperature of driest quarter	-0.441	0.662
Bio10	Mean temperature of warmest quarter	-0.923	0.322
Bio11	Mean temperature of coldest quarter	-0.695	0.697
TXx	Maximum value of daily maximum temperature	-0.852	0.095
TNn	Minimum value of daily minimum temperature	-0.392	0.853
SU	Number of summer days	-0.790	0.072
FD	Number of frost days	0.611	-0.722
Bio12	Annual precipitation	0.877	0.350
Bio13	Precipitation of wettest month	0.867	0.327
Bio14	Precipitation of driest month	0.915	0.270
Bio15	Precipitation seasonality	-0.289	0.048
Bio16	Precipitation of wettest quarter	0.860	0.364
Bio17	Precipitation of driest quarter	0.902	0.301
Bio18	Precipitation of warmest quarter	0.853	0.339
Bio19	Precipitation of coldest quarter	0.842	0.389
Rx5day	Maximum consecutive 5-day precipitation	0.842	0.279
CDD	Maximum number of consecutive dry days	-0.854	-0.114

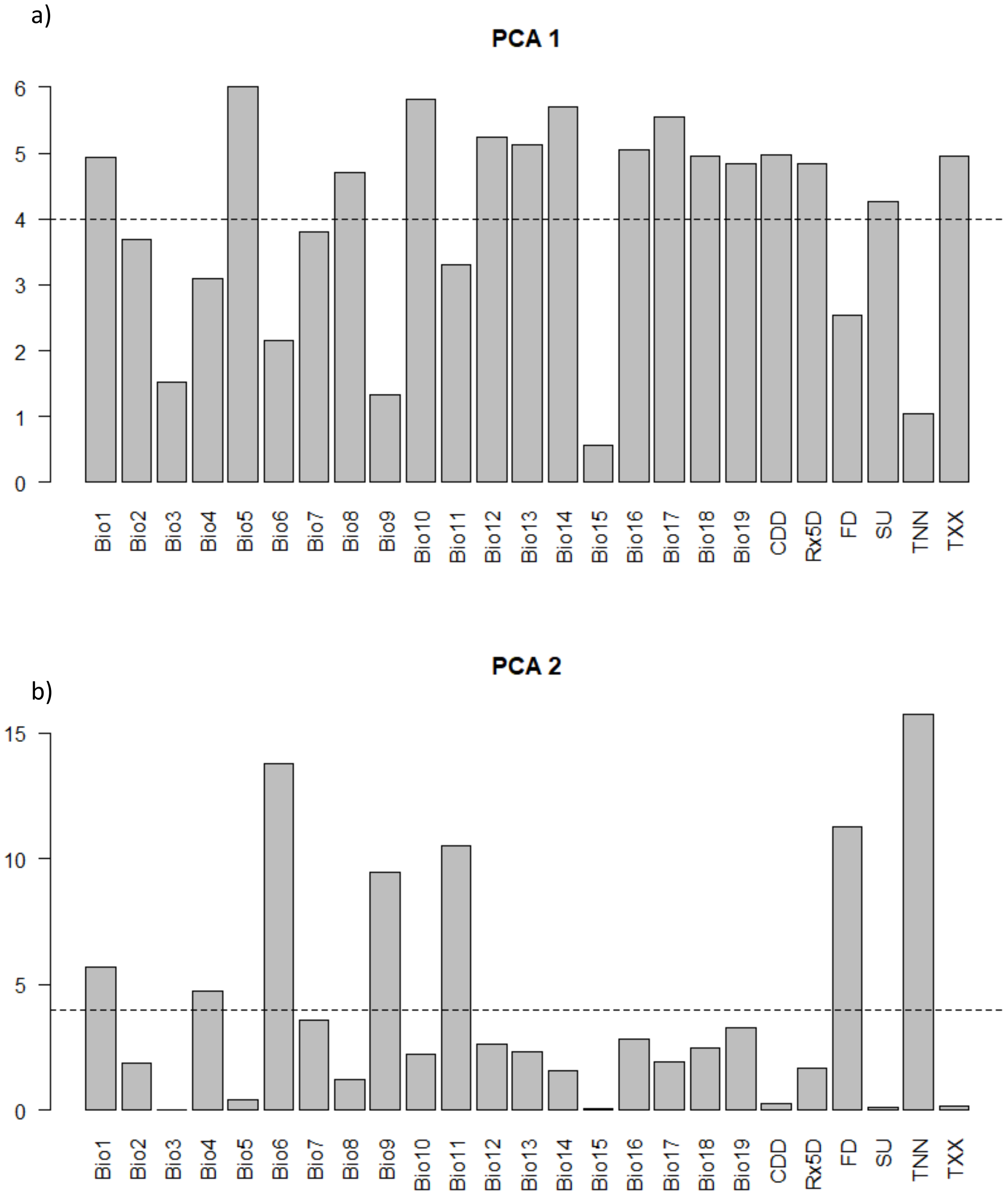


Figure 4.4. Contribution of the climatic variables to the first (a) and second (b) principal component axes. Contributions above 4 are greater than average (i.e. if contribution was equal among the 25 variables).

4.6.2. Gridded environmental space

PCA scores for the entire study area and each species occurrence during both time periods were subsequently projected into a grid of 1000x1000 cells bounded by the maximum and minimum PCA component scores across both periods – i.e., the gridded climate space. This methodology – calibrating the PCA with climate data from T1 and T2 combined and subsequently calculating species' PCA scores – means that the differences in position along the principal components discriminate differences between the T1 and T2 climate space, following previous temporal analyses (Ralston et al., 2017; Sillero et al., 2022; Tirozzi et al., 2022). A smooth kernel density function was applied to estimate the density of species occurrences in each cell of the climate space, correcting for the prevalence of the environments within the species range, as a measure to account for sampling bias and to ensure the results remain independent of the resolution of the climate grid chosen (Broennimann et al., 2011).

4.6.3. Niche dynamics

The niche analysis was conducted in R with the package ‘ecospat’ (Broennimann et al., 2023). To determine niche change over time, Schoener’s D index of niche overlap was calculated, chosen for its simplicity and long history of use (Warren et al., 2008). Schoener’s D, within the context of this investigation, provides a measurement of the overlap between the climatic niche experienced by each species at T1 and T2 ranging from 0 (no overlap) to 1 (complete overlap). This index is calculated by quantifying, for each grid cell in the climate space, the difference in smoothed kernel densities of species occurrence between periods. The niche comparison between periods was thereupon decomposed into three components to provide additional information about the drivers of niche change (Tirozzi et al., 2022): (i) niche expansion, the proportion of species’ niche present at T2 only; (ii) niche contraction (often termed niche unfilling), the proportion of species’ niche no longer present at T2; and (iii) niche stability, the proportion of niche present at both time periods.

4.6.4. Niche similarity test

Statistical tests for niche similarity between periods were used to determine whether observed climatic niches at T2 were statistically similar to the observed climatic niches at T1 using the ecospat.niche.similarity function within the ‘ecospat’ package. The hypothesis for niche conservatism was tested by comparing observed climatic niches at T1 with random simulations of species’ niches at T2 within the available climate space with the same kernel density distribution as T1, following the approach set out by Broennimann et al. (2011). By repeating these simulations 100 times, a null distribution of species overlap values were generated to determine if observed species niche overlaps are more statistically similar (niche conservatism) or not (niche divergence) than random, based on a significance threshold of 0.05.

4.6.5. Analogue climate

The overall approach – calculating niche overlap, determining niche dynamic indices and testing for niche conservatism through niche similarity testing – was first conducted including the full range of climatic conditions across both time periods. This was to provide a complete picture of the climatic niche changes that have occurred throughout the entirety of Great Britain, including novel climatic conditions and change between the first and the second time periods. This analysis was then repeated at the intersection of climates between the two time periods, termed the analogue climate analysis, to evaluate species niche dynamics considering only the climatic variability available during both periods thereby excluding niche shifts resulting from variation in the climatic space accessible to species (Tirozzi et al., 2022); subsequent results refer to this method unless specified otherwise.

4.7. EICAT/EICAT+ assessment

The potential impacts of species on the move with climate change are evaluated by employing frameworks from within invasion biology. The Environmental Impact Classification for Alien Taxa (EICAT) is an impact categorisation and scoring system whereby deleterious environmental consequences of alien taxa are assessed through their reported impact on native species as determined through a literature search. An EICAT+ assessment was subsequently developed by Vimercati et al. (2022) which is equivalent to EICAT but assesses potential positive impacts for native species. The EICAT and EICAT+ are implemented here to evaluate the potential negative and positive environmental consequences of species which are geographically tracking climate changes into new localities within the UK (Appendix D, table D.1).

4.7.1. Selection of study species

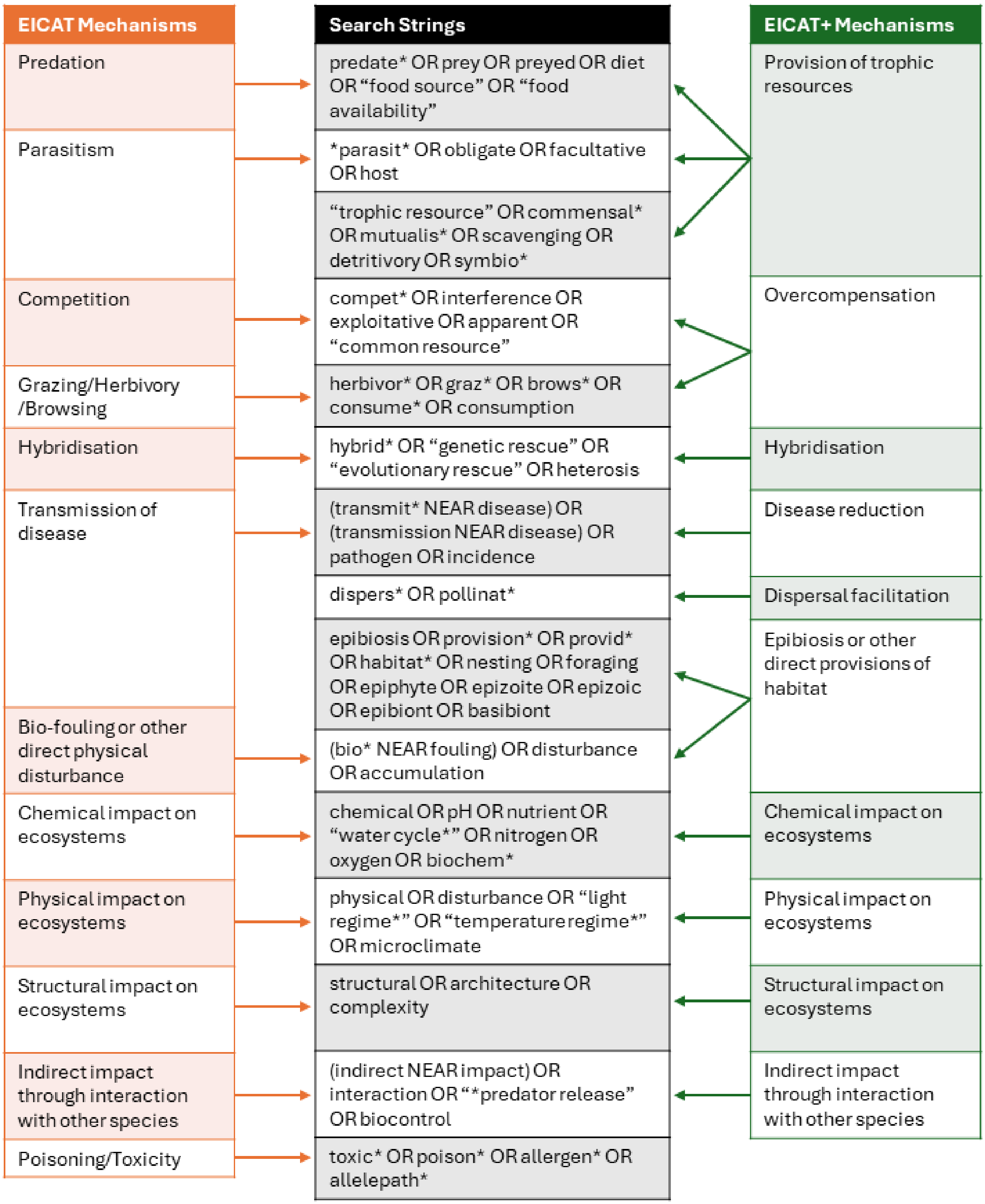
To select study species within the Odonata and Hymenoptera groups species, geographic distribution and realised climatic niche between periods T1 and T2 were compared. Species which increased in occupancy as well as exhibiting conservatism in their realised climatic niche between periods were included. Change in occupancy was determined by employing the method devised by Telfer et al. (2002; section 4.4.1.a) and species were tested for niche conservatism by testing for niche similarity between T1 and T2 (section 4.6.4). These species, having experienced geographic expansion during a period of recent warming whilst maintaining within a consistent climatic niche, are responding to climate change in a predictable way and likely to continue to arrive in new areas within Great Britain under continued climatic change. The study species are thus termed climate-tracking species, as they have been able to remain within their past climatic conditions as the climate has altered.

4.7.2. Literature search

(a) EICAT/EICAT+ mechanisms and search terms

The potential adverse impact of climate-tracking species was assessed by following the formal EICAT procedure outlined by the International Union for the Conservation of Nature (IUCN, 2020a; IUCN, 2020b). The potential positive impacts were assessed with reference to the EICAT+ framework developed by Vimercati et al. (2022) to complement that of EICAT. The EICAT/EICAT+ procedures involve a systematic review to gather observations of impacts by alien taxa to native taxa which are categorised under impact mechanisms. For EICAT there are twelve impact mechanisms including chemical, structural, and physical impacts on ecosystems as well as species interactions such as predation, competition, and parasitism (Van der Colff et al., 2020). EICAT+ includes ten impact mechanisms, some with sub-mechanisms, that either directly map onto EICAT (for example a species may have a deleterious impact by predating on native taxa or a positive impact through the provision of trophic resources by being prey for a native taxa) or are an additional mechanism, such as dispersal facilitation.

To implement the EICAT/EICAT+ protocol for the selected study species to assess potential impacts for other native species, required some minor adjustments. The EICAT guidance specifies that reviews for invasive species are to be exhaustive whereby the bibliographies and reference lists of initial search results are reviewed for potential additional sources of information with this process repeated until a point where no new data sources are identified (IUCN, 2020b). Moreover, the literature searches should include a range of online databases, web searches, and key texts encompassing published and unpublished literature (grey literature; IUCN, 2020b). Whereas here due to time restrictions the systematic literature review was conducted through a literature search on the Web of Science only, excluding other search platforms, grey literature, and further review of bibliographies within the literature retrieved. Furthermore, as conducted for native species, relevant literature was gathered by employing search strings specifically targeted for each of the impact mechanisms under EICAT and EICAT+, whereas searches for impacts of invasive species rely on search terms such as “introduced species”, “invasive species”, or “alien” to retrieve relevant information (ICUC, 2020b). Figure 4.5 details the search strings employed and how these map into the impact mechanisms of EICAT and EICAT+; literature was retrieved by searching for each species scientific and common names (table 4.3) alongside the targeted search strings (figure 4.5) with a separate search conducted for each mechanism. The results were then collated for each species and duplicates removed.



EICAT Mechanisms	Search Strings	EICAT+ Mechanisms
Predation	predate* OR prey OR preyed OR diet OR "food source" OR "food availability"	Provision of trophic resources
Parasitism	*parasit* OR obligate OR facultative OR host	
	"trophic resource" OR commensal* OR mutualis* OR scavenging OR detritivory OR symbio*	
Competition	compet* OR interference OR exploitative OR apparent OR "common resource"	Overcompensation
Grazing/Herbivory /Browsing	herbivor* OR graz* OR brows* OR consume* OR consumption	
Hybridisation	hybrid* OR "genetic rescue" OR "evolutionary rescue" OR heterosis	Hybridisation
Transmission of disease	(transmit* NEAR disease) OR (transmission NEAR disease) OR pathogen OR incidence	Disease reduction
	dispers* OR pollinat*	Dispersal facilitation
	epibiosis OR provision* OR provid* OR habitat* OR nesting OR foraging OR epiphyte OR epizoite OR epizoic OR epibiont OR basibiont	Epibiosis or other direct provisions of habitat
Bio-fouling or other direct physical disturbance	(bio* NEAR fouling) OR disturbance OR accumulation	
Chemical impact on ecosystems	chemical OR pH OR nutrient OR "water cycle*" OR nitrogen OR oxygen OR biochem*	Chemical impact on ecosystems
Physical impact on ecosystems	physical OR disturbance OR "light regime*" OR "temperature regime*" OR microclimate	Physical impact on ecosystems
Structural impact on ecosystems	structural OR architecture OR complexity	Structural impact on ecosystems
Indirect impact through interaction with other species	(indirect NEAR impact) OR interaction OR "*predator release" OR biocontrol	Indirect impact through interaction with other species
Poisoning/Toxicity	toxic* OR poison* OR allergen* OR allelepath*	

Figure 4.5. EICAT and EICAT+ impact mechanisms and search strings corresponding to each.

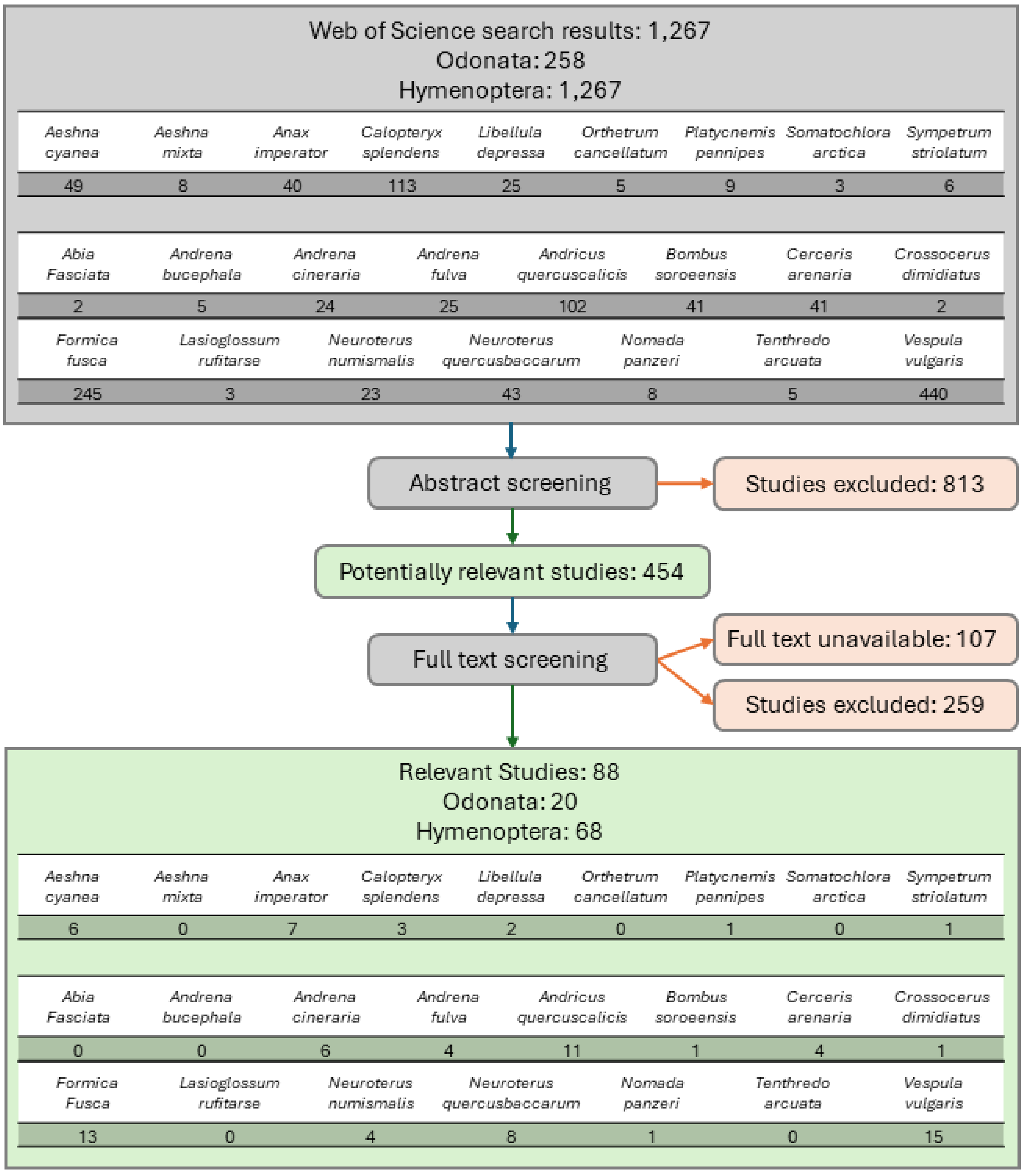
Table 4.3. Seach stings employed for each separate species.

Species	Common name	Search String
Aeshna cyanea	Southern Hawker	“aeshna cyanea” OR “southern hawker” OR “blue hawker” OR “southern aeshna”
Aeshna mixta	Migrant Hawker	“aeshna mixta” OR “migrant hawker”
Anax imperator	Emperor Dragonfly	“anax imperator” OR “emperor dragonfly” OR “blue emperor”
Calopteryx splendens	Banded Demoiselle	“calopteryx splendens” OR “banded demoiselle”
Libellula depressa	Broad-bodied Chaser	“libellula depressa” OR “broad-bodied chaser” OR “broad bodied chaser”
Orthetrum cancellatum	Black-tailed Skimmer	“orthetrum cancellatum” OR “black-tailed skimmer” OR “black tailed skimmer”
Platycnemis pennipes	White-legged Damselfly	“platycnemis pennipes” OR “white-legged damselfly” OR “white legged damselfly” OR “blue featherleg”
Somatochlora arctica	Northern Emerald	“somatochlora arctica” OR “northern emerald”
Sympetrum striolatum	Common Darter	“sympetrum striolatum” OR “common darter” OR “common sympetrum”
Abia fasciata	Banded Clubhorn	“abia fasciata” OR “banded clubhorn” OR “banded club-horn" OR “tenthredo fasciata” OR “zaraea fasciata”
Andrena bucephala	Big-Headed Mining Bee	“andrena bucephala” OR “big-headed mining bee” OR “big headed mining bee” OR “andrena longipes” OR “andrena eximia”
Andrena cineraria	Ashy Mining Bee	“andrena cineraria” OR “ashy mining bee”
Andrena fulva	Tawny Mining Bee	“andrena fulva” OR “tawny mining bee” OR “andrena armata” OR “apis fulva” OR “apis vestita”
Andricus quercuscalicis	Knopper Gall Wasp	“andricus quercuscalicis” OR “knopper oak gall” OR “knopper gall” OR “acorn gall”

Bombus soroeensis	Broken-belted Bumblebee	“bombus soroeensis” OR “broken-belted bumblebee” OR “broken belted bumblebee” OR “Ilfracombe bumblebee” OR “bombus soroensis” OR “seillean-mòr a' chrios bhriste” OR “seilleanan-mòra a' chrios bhriste”
Cerceris arenaria	Sand Tailed Digger Wasp	“cerceris arenaria” OR “sand tailed digger wasp”
Crossocerus dimidiatus	Blunt Tailed Digger Wasp	“crossocerus dimidiatus” OR “blunt tailed digger wasp” OR “crossocerus serripes”
Formica fusca	Silky Ant	“formica fusca” OR “silky ant” OR “dusky ant” OR “large black ant”
Lasioglossum rufitarse	Rufous-footed Furrow Bee	“lasioglossum rufitarse” OR “rufous-footed furrow bee” OR “rufous footed furrow bee” OR “lasioglossum atricorne” OR “lasioglossum rufitarse”
Neuroterus numismalis	Silk Button Gall Wasp	“neuroterus numismalis” OR “silk button gall” OR “silk-button gall” OR “silk button spangle” OR “silk-button spangle” OR “cynips numismalis”
Neuroterus quercusbaccarum	Spangle Gall Wasp	“neuroterus quercusbaccarum” OR “common spangle gall” OR “cynips quercusbaccarum”
Nomada panzer	Panzer's Nomad Bee	“nomada panzeri” OR “panzer’s nomad bee” OR “panzers nomad bee”
Tenthredo arcuate	Yellow-sided Clover Sawfly	“tenthredo arcuata” OR “common clover sawfly” OR “common clover-sawfly”
Vespula vulgaris	European Wasp	“vespula vulgaris” OR “common wasp” OR “common European yellowjacket” OR “common European yellow-jacket"

(b) Screening

Screening initial search results for relevance and therefore inclusion in the EICAT/EICAT+ assessments were undertaken through two stages. First articles were reviewed in terms of their title and abstract, and then were considered with reference to the full text if this was immediately and freely available in English. Therefore, articles which were not readily available in full were disregarded during the second stage of the screening process. If there was any uncertainty as to whether articles should be retained during the first stage, a precautionary approach was taken and these were retained and reviewed in full, following IUCN (2020b) protocol. The criteria for inclusion were that the article, i) was relevant to the topic, i.e. relating to the study species (climate-tracking Odonata or Hymenoptera) and documenting an impact through an EICAT/EICAT+ mechanism; and ii) had a relevant subject, i.e. that the impact documented was subject to another species native to the UK. For EICAT/EICAT+ impacts are required to have been reported within the same context under which they are being assessed (IUCN, 2020b), however, for this assessment observations reported from outside the UK were included provided they impacted a UK native species; such results were thereby assigned a confidence rating of 'low' due to the uncertainty resulting from spatial extrapolation. Figure 4.6 below documents the screening process.



Aeshna cyanea	*Aeshna mixta*	*Anax imperator*	*Calopteryx splendens*	*Libellula depressa*	*Orthetrum cancellatum*	*Platycnemis pennipes*	*Somatochlora arctica*	*Sympetrum striolatum*
49	8	40	113	25	5	9	3	6

Abia Fasciata	*Andrena bucephala*	*Andrena cineraria*	*Andrena fulva*	*Andricus quercuscalicis*	*Bombus soroeensis*	*Cerceris arenaria*	*Crossocerus dimidiatus*
2	5	24	25	102	41	41	2

Formica fusca	*Lasioglossum rufitarse*	*Neuroterus numismalis*	*Neuroterus quercusbaccarum*	*Nomada panzeri*	*Tenthredo arcuata*	*Vespula vulgaris*
245	3	23	43	8	5	440

Aeshna cyanea	*Aeshna mixta*	*Anax imperator*	*Calopteryx splendens*	*Libellula depressa*	*Orthetrum cancellatum*	*Platycnemis pennipes*	*Somatochlora arctica*	*Sympetrum striolatum*
6	0	7	3	2	0	1	0	1

Abia Fasciata	*Andrena bucephala*	*Andrena cineraria*	*Andrena fulva*	*Andricus quercuscalicis*	*Bombus soroeensis*	*Cerceris arenaria*	*Crossocerus dimidiatus*
0	0	6	4	11	1	4	1

Formica Fusca	*Lasioglossum rufitarse*	*Neuroterus numismalis*	*Neuroterus quercusbaccarum*	*Nomada panzeri*	*Tenthredo arcuata*	*Vespula vulgaris*
13	0	4	8	1	0	15

Figure 4.6. Summary of Web of Science search process and results for Odonata and Hymenoptera, indicating the total number of search results for each species and the total number of selected studies following the screening process.

4.7.3. Impact assessment

(a) Impact categories

Once the relevant studies have been screened for inclusion, each documented impact induced by each climate-tracking species on another UK native species under an impact mechanism must be assigned an impact category according to the EICAT/EICAT+ variables of interest (table 4.4 below). The variables of interest describe the change induced on other species in terms of measurable aspects with varying levels corresponding to impact severity. These are performance, change in population size, change in area of occupancy, and extinction status or extinction prevention. The impact category is assigned based on the severity of impact to recipient species from Moderate (MO; inducing a change in species' performance) to Massive (MV; causing irreversible community changes through local, sub-population or global extinction under EICAT or local re-establishment or extinction prevention of a population through EICAT+). Following the EICAT guidelines the impact can be either observed – directly based on documented impacts; or inferred – including assumptions made about the relationship between an observed variable to the variable of interest (IUCN, 2020b). Species which cause only a negligible level of impact or which induce no change in recipient species' performance are assigned the impact category of Minimal Concern (MC) or Minimal (ML+) positive impact depending on whether a negative or positive impact was investigated. The IUCN (2020b) guidelines provide five criteria under each mechanism through which to guide designation of an impact category. The methodology applied here followed these guidelines, with the only deviation being the assessment under the predation mechanism. For EICAT, predation of a native taxa is assigned to a minimum impact category of Minor (MN), even without evidence of a resulting decrease in performance of a native taxa, whereas under this assessment this limit was not employed ensuring a consistent evaluation for predation alongside other mechanisms. This approach guaranteed that impact severity was assessed equally across all impact mechanisms and verified that the impact of predation of native species by other climate-tracking natives was not inflated without appropriate evidence. Species overall impact is assigned as the most severe impact category ascribed, and in this case the greatest level of impact was also recorded separately for each of the mechanisms through which species impacted other native species.

Table 4.4. Categories for the Environmental Impact Classification for Alien Taxa (EICAT) and the Positive Environmental Impact Classification for Alien Taxa (EICAT+), including details for each severity level, shorthand title in brackets, and corresponding impact scores.

EICAT	EICAT+
Minimal Concern (MC) -1 Interacts with a species that can lead to negative impacts but causes only negligible level of impact	**Minimal positive impact (ML+) 1** Interacts with a species that can lead to positive impacts but causes only negligible level of impact
Minor impact (MN) -2 Causes a reduction in the performance of individuals	**Minor positive impact (MN+) 2** Causes an increase (or prevents decrease) in the performance of individuals
Moderate impact (MO) -3 Causes a decline in population size	**Moderate positive impact (MO+) 3** Causes an increase (or prevents decrease) in population size
Major impact (MR) -4 Causes a reversible decrease in area of occupancy through local or subpopulation extinction	**Major positive impact (MR) 4** Causes a transient increase (or prevents decrease) in species occupancy through local or subpopulation establishment
Massive impact (MV) -5 Causes irreversible community changes through local, subpopulation or global extinction	**Massive positive impact (MV) 5** Causes long-lasting increases (or prevents long-lasting decreases) in species occupancy through local or subpopulation reestablishment (or extinction prevention)

(b) Confidence rating

For both EICAT and EICAT+ a confidence rating is assigned which gives an indication of the degree of uncertainty attached to all impact classifications (IUCN, 2020b). Confidence is categorised into three levels – high, medium and low – after consideration of a number of different factors which influence the confidence in an assessment, following the IUCN (2020b) criterion. For the purposes of reviewing potential sources of uncertainty, here levels were assigned separately for each potential source of uncertainty and the overall lowest rating ascribed for each assessment. These sources are the presence of confounding effects, study design, data quality and type, spatial and temporal scale, and the coherence of evidence. Since this assessment included evidence from locations outside the UK, deviating from the EICAT guidance which recommends considering only impacts reported under the same context in which they are being assessed (IUCN, 2020b), an additional source of uncertainty was introduced by this investigation by extrapolating information from different contexts. Therefore, extrapolation was included as an additional category of uncertainty and marked as low confidence for evidence taken from outside the UK.

4.7.4. Spatial impact assessment

To assess the potential future impact of species on the move in Great Britain, the focus was on the positive and negative impact of species on other native taxa, when arriving in a new locality. Therefore, to map the impacts for both taxonomic groups (Odonata and Hymenoptera) the number of new climate-tracking species projected to arrive in each 5-km British National Grid was calculated, according to modelled current and future species presence (see section 4.5). The impacts of each species under each recorded impact mechanism were given a score based on the impact category prescribed; positive impacts assessed under EICAT+ were given positive scores ranging from 1 for a Minimal (ML+) positive impact to 5 for a Massive (MV+) positive impact and deleterious impacts assessed under EICAT were ascribed negative scores from –1 for Minimal Concern (MC) to –5 for Massive (MV) impact (see table 4.4 above). A spatial assessment of the impact of climate-tracking species could then be conducted by mapping species projected arrivals under a future climate scenario alongside the associated impact score for each impact mechanism for each species and totalling the impact score across all Odonatan and Hymenopteran species arrivals in each 5-km grid. Following this method, maps of potential species impacts were produced for both taxonomic groups, separately for positive and negative impacts, as well as for the net potential impacts resulting from both negative and positive impact mechanisms.

4.8. Statistical analyses

4.8.1. Comparing occurrence datasets

Pair-wise comparisons were carried out to compare predictions of Banded Demoiselle (*C. splendens*) distributions between models based on different occurrence datasets (traditional, citizen science, and social media) computed for both habitat suitability predictions and binary presence-absence maps (**chapter 5**). Similarity between predictions was calculated using Spearman’s correlation tests. Banded Demoiselle (*C. splendens*) habitat according to different data sources was further analysed by extracting the proportion of predicted presences within each of the 10 aggregate classes of the CEH 2015 land cover map (Land Cover Map 2015, 2017). This included a built-up areas and gardens class, to compare suitable habitat within urban areas across occurrence data types.

4.8.2. Odonata distribution and niche change

Average distances between central position at T1 and T2, as well as change in altitude, were compared between damselflies and dragonflies using a Mann-Whitney U test (**chapter 6**). The direction of shift for each species was analysed through a circular statistics approach conducted using the R package 'CircStats' (Lund & Agostinelli, 2018). A Watson two-sample test of homogeneity was calculated to investigate if there was any difference in the direction of species shifts between dragonflies and damselflies; this test is a non-parametric rank-based test that has been recommended for use by Landler et al. (2021) for comparing two samples of circular biological data. In addition, a Rayleigh test of uniformity was applied to determine whether species movements followed a northwards direction (i.e., with an alternative hypothesis specified at mean = 0°) and whether dragonflies or damselflies exhibit uniform distributions. The functions circ.mean, circ.disp, r.test, v0.test and watson.two within 'CircStats' were employed for this analysis. To determine whether there was any significant difference between dragonflies and damselflies in the proportion of species whose niche was conserved over time, a two-proportion Z-test was conducted in R.

4.8.3. Geographic versus climate niche shifts

Kendall correlation analyses were conducted to look for any significant relationship between geographic and climatic niche shifts of odonates (**chapter 6**); this type of analysis was deemed an appropriate non-parametric correlation test able to accommodate ties (Tirozzi et al., 2022). Correlation tests were run between species overlap and geographic distribution change, including distance of centroid shift and change in latitude and longitude of these distribution centres. Kendall correlation tests were also employed to determine whether there was any significant correlation between species occupancy trends and niche dynamics, including Schoener's D index of climate niche overlap, niche expansion, and niche contraction.

5. Investigating the potential of social media and citizen science data to track changes in species' distributions

5.1. Abstract

How to best track species as they rapidly alter their distributions in response to climate change has become a key scientific priority. Information on species distributions is derived from biological records, which tend to be primarily sourced from traditional recording schemes, but increasingly also by citizen science initiatives and social media platforms, with biological recording having become more accessible to the general public. To date, however, our understanding of the respective potential of social media and citizen science to complement the information gathered by traditional recording schemes remains limited, particularly when it comes to tracking species on the move with climate change. To address this gap, we investigated how species occurrence observations vary between different sources and to what extent traditional, citizen science, and social media records are complementary, using the Banded Demoiselle (*Calopteryx splendens*) in Britain as a case study. Banded Demoiselle occurrences were extracted from citizen science initiatives (iRecord and iNaturalist) and social media platforms (Facebook, Flickr, and Twitter), and compared with traditional records primarily sourced from the British Dragonfly Society. Our results showed that species presence maps differ between record types, with 61% of the citizen science, 58% of the traditional, and 49% of the social media observations being unique to that data type. Banded Demoiselle habitat suitability maps differed most according to traditional and social media projections, with traditional and citizen science being the most consistent. We conclude that (i) social media records provide insights into the Banded Demoiselle distribution and habitat preference that are different from, and complementary to, the insights gathered from traditional recording schemes and citizen science initiatives; (ii) predicted habitat suitability maps that ignore information from social media records can substantially underestimate (by over 3500 km^2 in the case of the Banded Demoiselle) potential suitable habitat availability.

5.2. Introduction

One of the swiftest consequences of climate change is the global redistribution of species on Earth (Scheffers et al., 2016; Pecl et al., 2017). Changes in the distribution of these species on the move are anticipated to have wide-reaching consequences for ecosystems and humans (Twiname et al., 2020; Wallingford et al., 2020). Consequently, how to best track these species as they rapidly alter their distributions has become a key scientific priority (Pecl et al., 2017). Information on species distributions is derived from biological records, which are defined as logs of species at a particular place at a certain time (Isaac & Pocock, 2015). Biological recording takes various forms and involves different contributors, methods, and information content. For a small number of taxa—namely those that are the most charismatic—structured monitoring schemes exist to provide systematic and focussed recording (Isaac et al., 2014). These include, for example for birds, the Breeding Birds Survey (Field & Gregory, 1999) and the Seabird Monitoring Programme (Walsh et al., 1995) in the UK, and the North American Breeding Bird Survey (Sauer et al., 1997). Such monitoring schemes are cost-intensive, requiring dedicated participants, typically involve standardized protocols (Isaac et al., 2014; Pocock et al., 2015) and tend to be biased toward more developed countries (Moussy et al., 2021). Most biological recording fits within opportunistic, unstructured recording schemes. These are generally coordinated by individual specialist recording schemes or societies that collate records with a particular taxonomic focus (Pocock et al., 2015).

With technological advancements making it easier to submit records, biological recording has become more accessible to the general public (Pocock et al., 2015). Several citizen science applications, such as iNaturalist, enable individuals to submit records that can be identified through the applications' community of scientists and naturalists (Nugent, 2018). Social media moreover offer a novel source of information for answering ecological questions about biodiversity, species distributions, and the impacts of climate change. Social media websites and applications allow users to post content of any kind, offering vast amounts of untapped, freely available information when this content is relevant to the ecological questions being investigated (see e.g., Allain, 2019; Barve, 2014; Daume, 2016; ElQadi et al., 2017; Pace et al., 2019). Yet, to date, our understanding of the potential of social media to complement existing sources of biological data for monitoring species distributions and habitat suitability availability remains limited, particularly when it comes to tracking species on the move with climate change (but see Pettorelli et al., 2019). In particular, information is lacking as to how species occurrence observations differ between different sources and to what extent different types of biological records are complementary.

To address this gap, this study makes use of available species occurrence data for the Banded Demoiselle (*Calopteryx splendens*) in Britain to assess the level of complementarity and divergence between distribution and habitat suitability maps derived from traditional recording schemes, citizen science initiatives, and social media information.

The Banded Demoiselle is a highly recognizable damselfly that is currently shifting its distribution in the UK due to climate change (Brooks et al., 2007; Mill et al., 2010; Cham et al., 2014; Pettorelli et al., 2019). It is a member of Odonata (dragonflies and damselflies), and as such has a hemimetabolous life cycle consisting of egg, nymph, and adult stages (Stoks & Córdoba-Aguilar, 2012). The nymphs are aquatic with eggs laid in aquatic plant tissue or in water, before metamorphosing into the terrestrial, flying adult stage, therefore requiring both healthy aquatic and resource-rich terrestrial habitats (Nagy et al., 2019). It is one of a few British riverine Odonates, requiring an adequate unidirectional flow for larval respiration, therefore restricted primarily to slow-flowing streams and rivers in lowland areas of southern Britain, although shifting further northward in recent years (Ward & Mill, 2005).

Britain makes for an excellent case study due to the vast availability of species distribution data for the UK, being arguably the most intensively recorded country on earth (Powney & Isaac, 2015), with the second greatest number of species occurrence records worldwide, behind the United States but with approximately eight times the record density (https://www.gbif.org/the-gbif-network, accessed April 2021). Odonata are a charismatic taxon, with a high engagement in recording both from volunteers within the UK's specialized recording scheme run by the British Dragonfly Society, as well as appealing to citizen-scientists more generally. The Banded Demoiselle, in particular, has a unique appearance and ease of species identification, being only one of two species of Demoiselle in the country with coloured wings (Svensson et al., 2004), making it an ideal candidate for investigation into the use of social media and citizen science occurrence records. Based on previous work (Dickinson et al., 2010; ElQadi et al., 2017; Callaghan et al., 2018; Noviello et al., 2021), we expect (H1) habitat suitability maps derived from social media records and citizen science initiatives to significantly differ from habitat suitability maps derived from traditional records and (H2) occurrences derived from social media platforms and citizen science initiatives to be more common in urban settings compared with traditional biological recording.

5.3. Methodology

5.3.1. Species occurrence data

Species occurrence records for the Banded Demoiselle were downloaded from both the Global Biodiversity Information Facility (GBIF.org, 2021) and the National Biodiversity Network (NBN) Atlas (British Dragonfly Society Recording Scheme, 2021; National Biodiversity Network Trust, 2021). Records were selected from 2010 onwards for comparison with social media datasets. Biological records from the British Dragonfly Society (BDS) Recording Scheme (excluding records from iRecord), Local Environmental Record Centres (LERC) as well as various national and international trusts and organizations were labelled as “traditional.” Records from both the iRecord and iNaturalist platforms were labelled as “citizen science.”

Records were collected from social media platforms (Facebook, Twitter, and Flickr) using the search terms “Banded Demoiselle” and “*Calopteryx splendens*.” For Twitter (table A.1) and Facebook (table A.2), this involved a manual search (completed between 13/01/2022 and 04/04/2022, for approximately 1.5 hours a day), with biological records consisting of an identifiable photograph or video. These records included either a tagged location or a mention of location within the content of the post, as well as a date for the observation if provided (otherwise the date the content was shared). Latitude and longitude information is generally preferable, allowing for precise placement of species occurrences. However, this information was not available for Twitter or Facebook records. Around 23% of the records found included a tagged location label; however, this was typically a city or town level. As such, records from Twitter and Facebook were manually checked and georeferenced by determining all the 1-km British National Grid squares that covered the spatial extent of the location description provided by the user. Although more imprecise than tagged geolocations, this ensured that the location information included was where the observation occurred (as opposed to where the photograph was uploaded). Searches yielded 95 results from Twitter and 331 from Facebook, which covered 295 and 867 1-km grid squares, respectively. These 1-km grid squares were included as Banded Demoiselle occurrences in subsequent Species Distribution Models (SDMs). For each social media occurrence, spatial precision (estimated to the nearest km^2) was recorded in the final dataset. For Flickr (table A.3), records were collated with the Flickr application programming interface (API) using the Flickr.photos.search (http://www.flickr.com/services/api/flickr.photos.search.html). Initial searches yielded 1316 results with location information as well as date recorded and posted that were extracted in R using the package 'FlickrAPI' (Ando & Pousson, 2022). These results were then manually verified, with 1223 observations remaining once records observed outside the relevant time frame or study location as well as irrelevant or misidentified observations were removed. For each data type, occurrence records were cleaned using the R package 'CoordinateCleaner' to flag and remove erroneous or duplicate results (Zizka et al., 2021). Potential data entry errors and failed georeferencing were flagged by checking for equal latitude and longitude values as well as zeros in the coordinates; coordinates matching country centroids and biodiversity institutions were also removed to ensure occurrences with imprecise georeferencing or captured individuals were excluded (Zizka et al., 2019).

5.3.2. Environmental data

The set of environmental variables considered to shape the distribution of Banded Demoiselle in the UK included climatic conditions, topography, landcover type, vegetation productivity, and level of urbanization. Monthly minimum and maximum temperature as well as monthly precipitation for the period 1990 to 2020 were accessed from the Met Office at a 1-km resolution (Met Office et al., 2022) and used to generate a series of monthly average bioclimate variables using the biovars function in the R package 'dismo' (Hijmans et al., 2021), under the assumption that species' ranges respond to the long-term averages of climate conditions (Taheri et al., 2020). These climate variables represent annual trends, seasonality, and limiting environmental factors and as such are designed to be biologically meaningful, being widely used for SDMs (Manzoor et al., 2018), and informative for Odonatan distributions (Collins et al., 2017; Abbott et al., 2022).

Slope was extracted from the Ordnance Survey (OS) Terrain 50 Digital Terrain Model (DTM) accessed from EDINA Digimap (OS Terrain 50, 2013); slope is important for Odonata species due to its influence velocity, O_2 on water content, weathering, channel substrate size, and organic matter composition (Collins & McIntyre, 2015) and of particular importance to the Banded Demoiselle that favours slow-flowing rivers.

To capture the aquatic element of the Banded Demoiselle's niche, the percentage cover at 1-km resolution of the freshwater aggregate class was extracted from the Centre for Ecology and Hydrology (CEH) 2015 Land Cover Map accessed from EDINA Digimap (Land Cover Map 2015, 2017). A Water and Wetness Probability Index (WWPI) product coordinated by European Environment Agency (EEA) Copernicus program was also acquired which indicates the occurrence of water and wet areas as a continuous probability at 20-m resolution based on observations between 2009 and 2015 (Langanke et al., 2018).

Normalized Difference Vegetation Index (NDVI; Pettorelli, 2013) Long Term Statistics (LTS) version 2.2. were also included from the Copernicus Global Land Service (CGLS) at a 1-km resolution (Toté et al., 2021). These statistics include the minimum, median, maximum, average, and standard deviation calculated from 10-daily NDVI values throughout 1999 to 2017 derived from Spot-4, Spot-5, and Proba-V satellite imagery. The NDVI gives an indication of “greenness” and therefore is likely to be influential in Odonatan distribution. In addition, the CGLS 100-m resolution tree cover density for the reference year 2012 was included (European Environment Agency, 2018). These should account for the influence of vegetation on the Banded Demoiselle distribution, where vegetation influences territory selection and where eggs are laid into aquatic emergent vegetation (Ward & Mill, 2005). To account for varying levels of urbanization, annual composites of visible night light version 2 were acquired for the years 2014 to 2018 from the Earth Observation Group (Elvidge et al., 2021) and averaged across these years.

Predictor variables were reprojected to the British National Grid and aggregated to a 1-km resolution where needed using the functions projectRaster and aggregate in R package ‘raster’ (Hijmans & Van Etten, 2012). All predictors were scaled to a mean of zero and a standard deviation of one. Predictor distributions were checked for any significant skew and a log transformation applied where a strong skew was identified.

The Pearson's correlation coefficient was calculated between each pair of predictor variables and where the coefficient was greater than 0.7, only one variable was retained. Including covarying predictors above this threshold results in increased uncertainty and disagreement among projections (Dormann et al., 2012; Brun et al., 2019). In cases where it was unclear which covarying predictor should be kept, two separate models were run with each set of covarying predictors, and the variable that contributed to more accurate model fit (assessed by True Skill Statistic [TSS] and the area under the receiver operating characteristic curve [AUC]) was kept. As a final check to ensure no correlated predictor variables were included, the Variable Inflation Factor (VIF), a measure of multicollinearity, was calculated for each occurrence dataset before model computation, to ensure that VIF was less than six, which is deemed acceptable (Guisan et al., 2017).

A preliminary set of SDMs was implemented through biomod2 with a dataset of all species occurrence records and all environmental variables to examine variable importance and guide predictor selection. Importance was determined by computing the Pearson's correlation between predictions made with a given variable and with the variable replaced with a randomized input, with variable importance averaged from five permutations. These preliminary screening steps resulted in a final set of predictors consisting of mean annual temperature, isothermality, mean temperature of the wettest and driest quarters, total annual precipitation, slope, percentage freshwater cover, WWPI, mean NDVI, and percentage tree cover.

5.3.3. Sampling effort

Species distribution models rely on the assumption that sampling effort and probability of detection are approximately even over a given area. However, this is often not the case, especially for opportunistically sampled data such as in citizen science projects and social media, and as such sampling bias can severely distort results (Bird et al., 2014; Johnston et al., 2021). A typical way to counteract this is with a target-group background approach (Phillips et al., 2009), which uses sampling from other related taxonomic groups to give a broad overview of sampling effort over an area. In this study, this approach was not possible as acquiring an equivalent sampling background for social media data is extremely difficult, if not impossible, due to the time and computational workload involved. Instead, we used a 'bias covariate correction' method (Warton et al., 2013; Chauvier et al., 2021), where several proxies for sampling effort are used to correct for areas of bias. We therefore included several sampling effort predictors in our models, namely distance to major population centre, distance to nearest road, and population density. Shapefiles for major population centre were downloaded from the Office for National Statistics (2021) and the Scottish Government SpatialData.gov.scot (2022), and the distance from each 1-km grid cell in our study area to the nearest city was calculated.

Spatial line data for roads were based on OpenStreetMap Data Extracts, as processed by Geofabrik GmbH (2023), using the latest road data available for the UK as of February 13, 2023; for each grid cell in the study area, we calculated how far they lay from the nearest road. Residential population density was downloaded from the Environmental Information Data Centre (2023) at 1-km resolution. Predictor covariation was assessed, and a preliminary set of models was run to check for variable importance (following same methods as for environmental variables). Where sampling effort variables were important ($1 - r > 0.05$, where r is the Pearson's correlation coefficient), they were retained in the final model. When final projections were made, these variables were set to the median value for a given layer across the study area, to compensate for the potential effect of sampling effort following the protocol of Warton et al. (2013).

5.3.4. Species distribution modelling

Ensemble SDMs for the Banded Demoiselle were implemented using the R 'biomod2' package (Thuiller et al., 2021) for each species occurrence dataset. There was no a priori reason to select one family of models over another, so all were trialled and compared in terms of habitat suitability outputs, performance metrics provided by biomod2 (accuracy, bias, TSS, and AUC), and variance in estimated response curves. Since all performed similarly and showed broadly similar outputs, ensemble model results were built with output from all high-performing models, regardless of family. As such, a set of six modelling techniques were selected including three machine-learning methods, generalized boosting model, random forest, and maximum entropy; two regression methods, generalized linear model, and multiple adaptive regression splines; and finally, a recursive partitioning method, classification tree analysis. For all modelling algorithms, the default biomod2 settings were used.

For each dataset (traditional, citizen science, and social media), 10,000 pseudo-absence points were randomly selected from the background data, a quantity approximately matching the most numerous occurrence dataset, to be broadly appropriate across SDM techniques (Barbet-Massin et al., 2012). To ensure pseudo-absence composition was not impacting results, preliminary SDMs were computed with 5, 10 and 15 sets of pseudo-absence points. Agreement was high overall across all statistical metrics used and did not differ significantly between runs with different numbers of pseudo-absence sets. As each dataset was large and computationally taxing, all final models were run with five pseudo-absence datasets. To reduce the potential of selecting pseudo-absences within the same niche as presences, pseudo-absences were placed at least 1.5 km away from any observed occurrences that have a coordinate uncertainty of up to 1 km.

Several validation models were created, where 20% of the species occurrences, including both presence and pseudo-absence points, were set aside for evaluation. Model performance was assessed with TSS, which provides a threshold-independent measure of accuracy (Allouche et al., 2006). TSS has several documented drawbacks (Leroy et al., 2018), notably its dependence on prevalence; however, we chose to use a balanced approach where the number of pseudo-absences was set to match the number of presences, as this reduces the chance of bias when using TSS results, allows easier comparison between different models as prevalence is held constant, and is the recommended approach when attempting to maximize discrimination in SDMs (Steen et al., 2020). Several alternative metrics were also calculated to provide an overall summary of performance and potential bias. These included AUC; frequency bias, the ratio between observed and predicted presences; accuracy, the fraction of occurrences correct; and finally Cohen's Kappa coefficient, a measure of model accuracy which corrects for accuracy expected to occur by chance (Allouche et al., 2006). This process was repeated five times, splitting the occurrences into five random training and testing sets of 80% and 20%, respectively, balancing the ratio of presence and pseudo-absence points, to ensure that their composition was not having any impact on model accuracy. Ensemble models were built combining all individual models with a TSS value greater than 0.6, considered to be useful to excellent (Komac et al., 2016), and weighing model contribution according to their TSS.

The evaluation results are based on the internally validated models, whereas the final projections presented throughout the manuscript are based upon all available occurrence data, without any presences or pseudo-absences set aside for internal validation. This is to ensure the final parameter estimates are built with the maximum information and therefore lower uncertainty in parameter estimates and projections. As our validation models were robust, we verified that the final full models were sufficiently similar to the validation models so as to ensure the final full models were similarly robust. We verified this using a Spearman's correlation between the projected habitat suitability of five validation models and the final models for each data source.

Each ensemble model of habitat suitability was converted into binary presence-absence maps; thresholds were selected to maximize the combined sensitivity and specificity scores (Liu et al., 2016). Pair-wise comparisons were carried out to compare predictions between models based on different occurrence datasets, computed for both habitat suitability predictions and binary presence-absence maps. Similarity between predictions was calculated using Spearman's correlation tests.

Banded Demoiselle habitat was further analysed by extracting the proportion of predicted presences within each of the 10 aggregate classes of the CEH 2015 land cover map accessed from EDINA Digimap (Land Cover Map 2015, 2017). This included a built-up areas and gardens class, to compare suitable habitat within urban areas across occurrence data types.

5.4. Results

A total of 17,831 observations of the Banded Demoiselle were collected (table 5.1). When gridded to the 1 km^2 British National Grid, at the same resolution as the predictor variables, a large proportion of the total number of grid cells where presence was reported for each occurrence type, were unique to that data type; ~61%, ~58% and ~49% for citizen science, traditional and social media, respectively. When aggregated to 10 km^2, the difference becomes less stark (table 5.1; figure 5.1).

Table 5.1. Total number of occurrence records collected for each type, including the number of 1 km and 10 km British National Grid squares. For each type, the number and proportion of grid squares where observations were unique to that type is given. Abbreviations: BDS, British Dragonfly Society; LERC, Local Environmental Records Centre.

Type	Details	Total records	1 km grids	Unique 1 km grids	10 km grids	Unique 10 km grids
Traditional	BDS; LERCs; National Trusts	6,749	4,211	2,424 (57.6%)	908	184 (20.3%)
Citizen Science	iRecord; iNaturalist	9,646	5,075	3,100 (61.1%)	982	136 (13.8%)
Social Media	Facebook; Flickr; Twitter	2,026	1,480	726 (49.1%)	421	15 (3.6%)

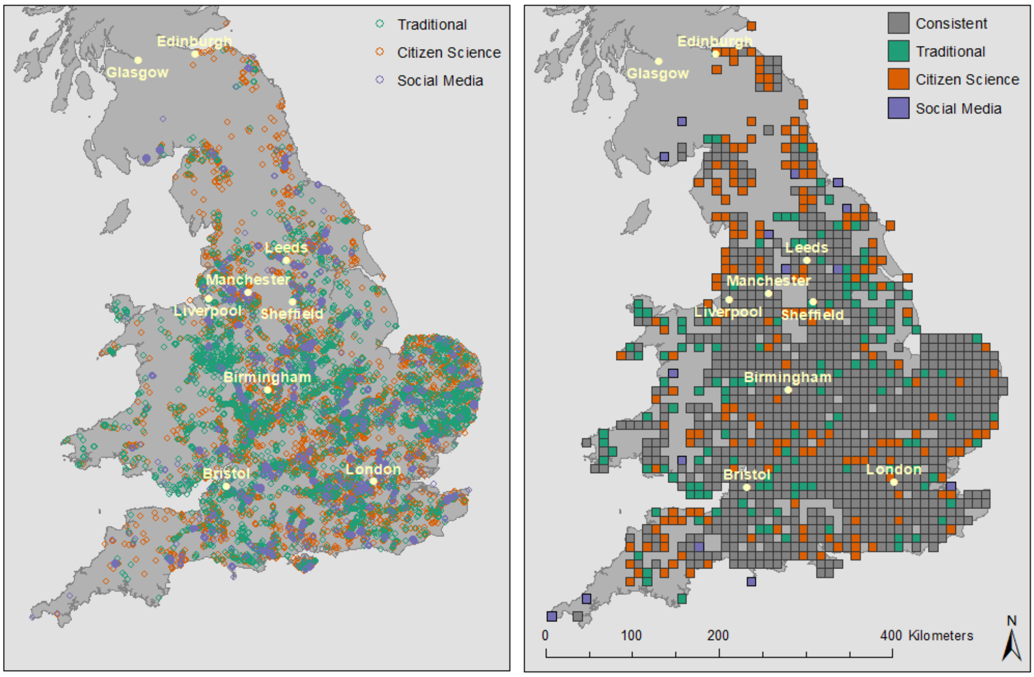


Figure 5.1. Distribution of traditional, citizen science, and social media species occurrence records (left) and consistencies and differences when gridded to the 10 km British National Grid (right). Population centres with more than 500,000 people have been highlighted.

The TSS and Kappa scores across all SDMs were greater than 0.6, while all AUC values exceeded 0.85, indicating good model performance (table 5.2). Model performance was broadly similar across all data sources (table 5.2). Accuracy and bias values were similar across data types, and high across all models. Validation models were representative of the final models as Spearman's correlation coefficients between validation and final models were greater than 0.98 in all cases.

Table 5.2. Evaluation statistics for the ensemble models averaged from validation runs for each species occurrence data type, including the True Skill Statistic (TSS), the Area Under the Receiver Operating Characteristic (ROC) Curve (AUC), Cohen's Kappa coefficient, sensitivity, and specificity. Values in brackets are the standard deviation across the five validation runs.

Occurrence dataset	TSS	AUC	Kappa	Accuracy	Bias
Traditional	0.60 (0.05)	0.88 (0.03)	0.60 (0.05)	0.80 (0.02)	0.99 (<0.01)
Citizen science	0.66 (0.05)	0.91 (0.02)	0.65 (0.04)	0.84 (0.02)	0.99 (<0.01)
Social media	0.66 (0.04)	0.90 (0.02)	0.62 (0.05)	0.86 (0.02)	0.99 (0.02)
All	0.65 (0.05)	0.90 (0.02)	0.61 (0.05)	0.87 (0.02)	1.00 (<0.01)

Annual mean temperature and percentage freshwater cover were highly ranked variables for all three data sources and were found to be important in all three models ($1 - r > 0.1$, where r is the Pearson's correlation coefficient). In addition, summed annual precipitation was found to be highly important in citizen science and traditional SDMs, but not for social media. Distance to the nearest roads was an important predictor for social media SDMs but was less important when using traditional or citizen science data sets. The breadth of suitable environmental conditions and response curves were broadly similar across data types.

Distance to roads was the only covariate of sampling effort that was found to have any effect on the models, and outputs shown here are made following correction for sampling effort. Comparisons with uncorrected models indicated significant differences in suitability for social media SDMs can be seen around major population centres including London, Manchester, and Birmingham.

Social media had higher spatial uncertainty than data from other sources, so several sensitivity tests were carried out. SDMs were constructed with points with a spatial precision of at least 1, 2, 5, 10 km^2, respectively, and compared to models constructed with the full data set. The most dissimilar models were those built with all data and those built with 2 and 1 km^2 precision data (Spearman's coefficient: 0.96 and 0.97 respectively). All models were broadly similar, though uncertainty was higher around major population centres and coastal areas. The results presented here are for models built with all data.

Under our ensemble model based on traditional occurrence records, around 50,800 km^2 (21.71%) of Great Britain's landmass is predicted suitable for the Banded Demoiselle; this is compared to ~54,600 km^2 (23.33%) based on citizen science records and ~41,500 km^2 (17.73%) based on social media records (figure 5.2). As expected, using all collected data led to the greatest total projected area of suitable habitats for the Banded Demoiselle (~57,600 km^2, 24.60%). Suitable habitats for the Banded Demoiselle were predicted to primarily include arable lands (37.9% to 48.5% of total suitable area), improved grasslands (32.6% to 33.5%) and built-up areas (11.8% to 21.0%), with only a small proportion of suitable areas found within broadleaf woodlands (3.1% to 3.8%). The study area was similarly dominated by the arable and improved grasslands land cover types, covering together 57.6% of the total area (table 5.3).

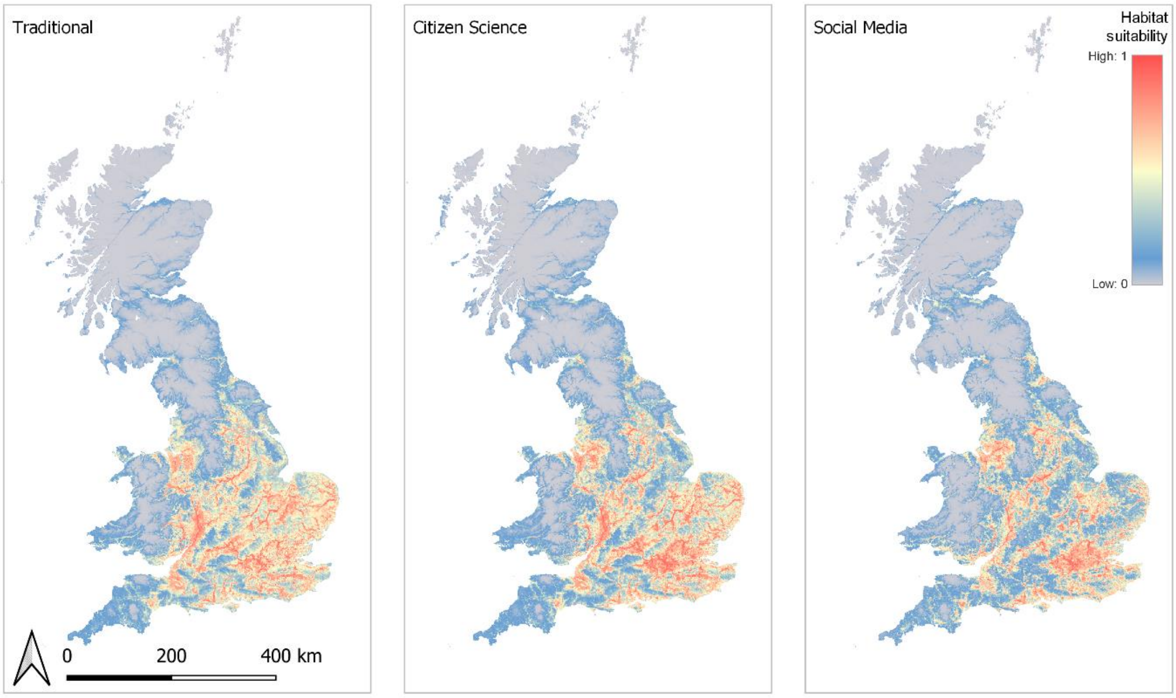


Figure 5.2. Projected habitat suitability index according to weighted mean ensemble models computed based on traditional (left), citizen science (middle) and social media (right) observations.

Table 5.3. Coverage of land cover classes for the Great Britain study area and the predicted suitable habitat for the Banded Demoiselle according to ensemble species distribution models based on different types of species occurrence records. Percentages are given of total study area and total predicted suitable habitat, with values in brackets being the total area in kilometres squared. Bold text is used to indicate land classes where Banded Demoiselle suitable habitat dominates (where total suitable area >1000 km^2).

Class	Study Area	Traditional	Citizen Science	Social Media
Improved grassland	**31.2% (73,084)**	**33.5% (17,003)**	**32.6% (17,770)**	**33.4% (13,854)**
Arable	**26.4% (61,865)**	**48.5% (24,636)**	**47% (25,642)**	**37.9% (15,747)**
Mountain, heath, bog	15.4% (35,926)	0.4% (195)	0.4% (244)	0.4% (181)
Semi-natural grassland	9.5% (22,113)	0.7% (339)	0.6% (351)	0.8% (334)
Built-up areas and gardens	**6.6% (15,394)**	**11.8% (6,004)**	**13.7% (7,455)**	**21% (8,716)**
Coniferous woodland	6.1% (14,303)	1.0% (502)	1.1% (574)	1.0% (400)
Broadleaf woodland	**2.5% (5,919)**	**3.1% (1,552)**	**3.4% (1,850)**	**3.8% (1,571)**
Coastal	1.2% (2,831)	0.5% (230)	0.5% (290)	0.7% (284)
Freshwater	0.6% (1,512)	0.6% (321)	0.7% (372)	0.9% (372)
Saltwater	0.4% (1,042)	0.0% (24)	0.1% (36)	0.1% (36)

Spearman's correlation coefficients between habitat suitability maps based on different record types were greater than 0.85 for all pairs of occurrence datasets. Projections based on traditional and citizen science records were the most correlated (0.95) while projections based on traditional and social media records were the least correlated (0.87, table 5.4). The area consistently expected to be suitable for the Banded Demoiselle was estimated to cover 44,761 km^2 when comparing models based on traditional and citizen science records; but this area was expected to only cover 33,061 km^2 when comparing models based on traditional and social media records. In the latter situation, 17,745 km^2 of suitable habitats was uniquely identified by traditional records while 8,434 km^2 of suitable habitats was uniquely identified by social media records. The area uniquely identified as suitable by traditional records primarily covers the southern lowlands, while the area uniquely identified as suitable by social media records covers the southwest, south Wales, coastal areas around the south of the UK, the northeast and Scotland (figure 5.3). A greater proportion of projected suitable habitat was found within built-up and urban areas when considering social media records (21%) than citizen science (13.7%) and traditional data (11.8%).

Table 5.4. Spearman's correlation between models derived from different species occurrence records. Above diagonal values are the correlation between binary presence-absence maps and below diagonal the correlation between habitat suitability projections.

	Traditional	**Citizen Science**	**Social Media**
Traditional	1	0.805	0.651
Citizen Science	0.952	1	0.714
Social Media	0.870	0.928	1

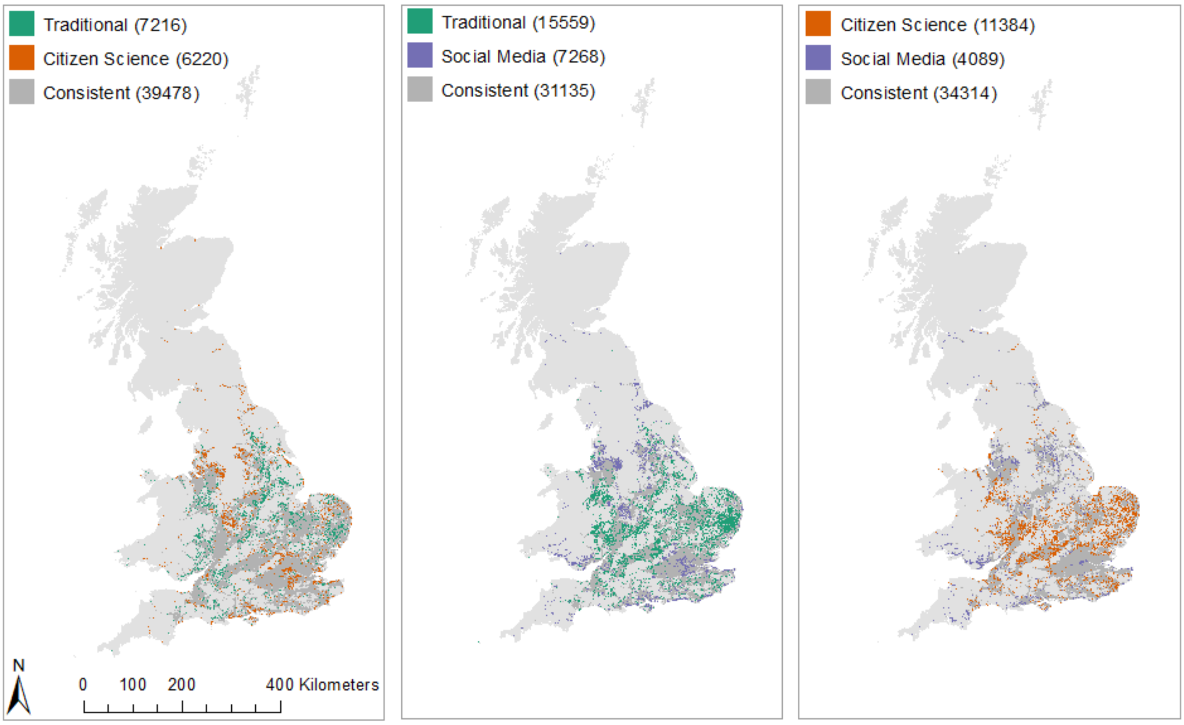


Figure 5.3. Pairwise comparison between projected suitable habitat for the Banded Demoiselle according to different data types. Predictions were converted to binary presence–absence maps using the threshold that maximized the true skill statistic for each ensemble model. Values in brackets indicate the total consistencies and differences between predicted suitable habitats in terms of the number of 1-km pixels and therefore total area in km^2.

5.5. Discussion

This study offers a unique assessment of the level of complementarity and divergence between habitat suitability distributions derived from traditional recording schemes, citizen science initiatives, and social media information. Our results show that (i) social media records provide insights into the Banded Demoiselle distribution and habitat preference that are different from, and complementary to, the insights gathered from traditional recording schemes and citizen science initiatives; (ii) predicted habitat suitability maps that ignore information from social media records substantially underestimate (by over 3500 km^2) potential suitable habitat availability.

The use of social media to extract species occurrence observations and inform ecological research and wildlife management is a relatively new concept, with a few cases where such methods have been investigated both for native (Barve, 2014; ElQadi et al., 2017; Pace et al., 2019) and non-native species (Allain, 2019; Daume, 2016). Social media data can greatly extend the number of occurrence records available to ecologists (Allain, 2019) and, in the case of countries with limited resources, provide an alternative to costly specialized recording schemes and citizen science campaigns (Di Minin et al., 2015). Our study demonstrates that there is much value in capitalizing on this new type of information: Even though substantially less numerous than the other data types overall, 49% of the Banded Demoiselle presences derived from social media platforms were unique to social media, enabling us to capture a broader perspective on the species' habitat preferences. Our conclusions resonate with previous research by ElQadi et al. (2017) who found that Flickr observations of honeybees in Australia (i) extended the known distribution based on traditional records towards urban centres, and (ii) represented tourist areas in remote locations that were not depicted by traditional records (ElQadi et al., 2017). Together, these results suggest that spatial patterns in social media recorder activity tend to be different from the patterns found among recorders involved with traditional and citizen science data collection.

Our findings demonstrate that social media projections of Banded Demoiselle habitat cover a larger proportion of built-up areas and gardens than traditional recording. This may potentially be an artifact of sampling bias, but it may also indicate that these urban areas provide important habitats for Banded Demoiselles, something that could be underestimated without the consideration of social media observations. The proportions of the other land cover types were largely consistent between data types, with predicted Banded Demoiselle habitat dominated by arable and improved grasslands. This contradicts previous findings that agriculture, managed land, and excessive grazing do not provide suitable Banded Demoiselle habitat due to diminished bankside vegetation (Lowdon, 2015; Ward & Mill, 2005). The coarse spatial resolution considered in this study, together with the fact that our study area is heavily dominated by these landcover types (covering 57.6% of our study area), may explain such results.

Sourcing information on species presence from social media platforms is not straightforward, and the amount of information garnered can be quite limited. For example, the manual Facebook and Twitter searches yielded 331 and 95 results, respectively, for Banded Demoiselle. These numbers are comparable with similar studies that have extracted species occurrence records from Facebook, such as the ones by (i) Campbell and Engelbrecht (2018) that gathered 1239 observations for 34 species of baboon spiders across Southern Africa (around 36 records per species), (ii) Rocha et al. (2017) that sourced 369 records of the Eurasian red squirrel in Portugal, and (iii) Havlin et al. (2017) that collected

30 observations of red-necked wallabies on the Isle of Man, UK. These investigations all used specific Facebook pages set up by the scientists and dedicated to encouraging submission of records for their target species. In our case, biological records were gathered from existing platforms, which may partially explain the low numbers of records we were able to source. Although requiring greater effort and longer term management, dedicated pages may yield a greater number of results as well as being a more active way of engaging communities with biological recording.

Acquiring biological records from Flickr was aided by the use of an API that allows for an automated search of visual content and extraction of information on associated location and date. Using this API for the Banded Demoiselle yielded 1316 initial results instantly, providing both a faster method to access information in comparison to other social media platforms investigated as well as yielding a greater total number of observations. Although the initial search was rapid in comparison with manual searches on Twitter and Facebook, the subsequent manual verification of the data was, however, time-consuming. The R package ‘CoordinateCleaner’ (Zizka et al., 2021) provided a means to rapidly flag and remove likely erroneous records, such as those assigned to country centres and biodiversity facilities, as well as identify outliers and duplicate observations. The difficulty with Flickr API searches is that this can yield observations where species are incorrectly identified, alongside content where the species name is mentioned in another context without any intention to indicate presence of the species. This verification step was proven to be important in our case, leading to the removal of 92 sightings (~7% initial results) despite the deliberate selection of an easily identifiable species. For other species, results may be even less reliable, such as for two bumblebee species in Australia where only 65% and 68% of the occurrences extracted from Flickr by ElQadi et al. (2017) were correctly identified. Research to identify alternatives to manual verification process is needed (ElQadi et al., 2017).

Citizen science has become an invaluable and cost-effective source of species occurrence records (Noviello et al., 2021). Nevertheless, a number of concerns remain about the accuracy and quality of citizen science data due to variability in volunteers' level of experience and expertise (Aceves-Bueno et al., 2017), with previous studies finding a lower performance of SDMs based on citizen science data compared with systematic surveys (Tiago et al., 2017) and suggesting filtering citizen science data according to data quality and information content for more accurate SDMs (Van Eupen et al., 2021). In our case, however, all SDMs performed adequately, and habitat suitability maps derived from traditional and citizen science sources were the most congruent. These comparable results from citizen science and traditional observations are likely partially a result of improved data validation within citizen science initiatives (Dickinson et al., 2010), with, for example, iNaturalist crowdsourcing verification from users within the platform and iRecord verification largely being performed by volunteers associated with national recording schemes, such as within the BDS—likely the same county recorders that oversee and verify the BDS's own records. Moreover, both the BDS and citizen science records are largely collected with an unstructured and opportunistic framework.

A number of limitations to our study should, however, be acknowledged. First, this work was performed at a relatively coarse resolution; fine-scale and more sophisticated hydrological and hydraulic predictor variables could prove advantageous for Odonatan modelling (Collins & McIntyre, 2015). Second, modelling approaches focused on rivers and water bodies, as opposed to approaches based on gridded variables as well as the combination of stream-only and terrestrial-only model processes, have been previously encouraged when aiming at identifying suitable habitats for freshwater species such as Banded Demoiselle (Collins & McIntyre, 2015). However, such an approach was not feasible here, particularly as the vast majority of occurrences collated were for the terrestrial adults as opposed to aquatic nymphs. Third, biotic variables have been increasingly employed to improve predictive ability of SDMs (Yates et al., 2018), with competition and intraguild predation particularly significant constraints on Odonata distributions (Pélissié et al., 2022); however, inclusion of these interactions as predictors for Banded Demoiselle habitat was beyond the scope of this study due to the quantity of interactions possible. As such, these biotic factors are likely to modify the projected potential suitable habitat throughout Britain in practice. Fourth, most of the Twitter occurrences lacked geo-location information and so, along with Facebook, relied on location information within the content that lacked precision compared with traditional occurrences. In this study, there was little evidence that using lower precision data significantly affected results, verified through several sensitivity analyses, but this is unlikely to be universally true and should be treated carefully. Fifth, for social media, when the location of the observation was not explicitly detailed an assumption was made that the tagged location provided information as to where the picture was taken; this cannot be confirmed and therefore adds a level of uncertainty regarding the reliability of social media data. Sixth, it is possible that individuals could report Banded Demoiselle occurrences with multiple sources, leading to duplicates that may affect the correlation and similarities between data types. Seventh, we found evidence that sampling bias can be more prevalent in citizen science and social media data, than in more traditional sampling surveys. There are numerous published methods of compensating for these issues (Stolar & Nielsen, 2014; Ranc et al., 2016; Chauvier et al., 2021), some of which were used here, but established methods may be difficult to carry out for limited social media data. Finally, while providing a compelling case for employing social media data for the Banded Demoiselle, the generality of our conclusions requires further investigation to determine whether our findings apply for other species, particularly those that are perhaps more difficult to identify by nonexperts.

5.6. Conclusion

Public participation has become commonplace within scientific research aimed at biodiversity monitoring and conservation, enabling access to a monumental breadth of data on species occurrence unobtainable otherwise. Our study offers a compelling illustration of the value of alternative sources of traditional biological records and highlights, in particular, the value of ecological information derived from social media data as an inexpensive and complementary source of species occurrence data. This source of freely available information can be exploited to capture a more complete understanding of species habitat preferences, appreciate the influence of urban settings, and gain insights that cannot be attained from traditional recording alone. We believe further development of APIs to gather social media information, technologies for automated verification, and greater adoption of available geo-tagging facilities, would further broaden the scientific application of social media.

6. Investigating odonates’ response to climate change in Great Britain: a tale of two strategies

6.1. Abstract

Aim: Species are largely thought to maintain broadly static niches over time, an assumption underpinning much theoretical ecology including the implementation of ecological models to project species' current and future distributions. Here, we assess niche conservatism in odonates in Great Britain over the past six decades by simultaneously quantifying changes in species geographic distribution and evaluating temporal trends in species realised climatic niche.

Location: Great Britain

Methods: Distributional changes were assessed by calculating changes in species distribution centres and deriving occupancy trends. Changes in climatic niches were assessed using a principal component analysis to quantify niche overlap, using information on both climate averages and extremes.

Results: We show that dragonflies and damselflies displayed distinct responses to changing climatic conditions. Dragonflies shifting to higher latitudes maintained, on average, greater consistency in their climatic niches, providing evidence for climate tracking. Greater climate niche flexibility and increased occupancy over time, on the other hand, were more common in damselflies.

Main Conclusions: We unveil evidence for climatic niche divergence in damselflies on a national scale, casting doubt on the relevance of species distribution models for predicting the impacts of climate change on this, and potentially other, groups of species. More broadly, our results call for more multi-species temporal comparisons of spatial distributions and climate niches during recent periods of changes in climatic conditions to improve our ability to contrast species' vulnerability risk to the ongoing climate crisis.

6.2. Introduction

Species redistribution is one of the swiftest responses to rapid changes in climatic conditions (Scheffers et al., 2016). As climate changes, species often persist by shifting their geographic range across the landscape to track their ecological niche. The study of the factors underpinning species redistribution is a rapidly developing research area (Bonebrake et al., 2017), with an abundance of evidence that species have already shifted their geographic ranges in response to recent anthropogenic climate change, with potentially profound social and economic impacts (Pecl et al., 2017).

Opportunities for range shifts are known to be influenced by many factors in addition to climate, such as species interactions, dispersal, land use, topography and landscape connectivity. Consequently, many species may not be able to keep pace with climate change, struggling to disperse rapidly enough (Jezkova & Wiens, 2016) or to navigate our increasingly human-modified landscapes. Posed with such challenges, these species may instead shift their ecological niche whilst maintaining a stable geographic distribution – as opposed to maintaining a stable realised niche in environmental space and shifting their geographic range (Wang et al., 2023).

Historically, niche conservatism – the tendency of a species' niche to remain unchanged over time (Pearman et al., 2008) – has largely been assumed when investigating or predicting species biogeographical patterns and informs much theoretical ecology, underpinning in particular ecological niche modelling efforts (Guisan et al., 2014; Jezkova & Wiens, 2016). Nevertheless, studies investigating niche conservatism have found disparate results (Guisan et al., 2014), with evidence suggesting that species' niches can shift across different environments or timescales, rather than remaining static (Jezkova & Wiens, 2016).

Temporal comparisons of climate niches across sets of related species during periods of rapid changes in climatic conditions could represent an effective way to explore how common shifts in species' niche are. This approach, however, has received little attention to date (Lustenhouwer & Parker, 2022), having been largely confined to plants (Wang et al., 2023) and birds (Ralston et al., 2017; Nadeau & Urban, 2019). Comparing responses across species enables researchers to identify the factors shaping species' ability to track or adapt to climate change, which could prove crucial for determining and predicting species risk to ongoing, rapid anthropogenic climate change.

To address this knowledge gap, we investigate species' geographic and climate niche shifts in Great Britain's odonates (dragonflies and damselflies). Great Britain offers a wealth of biological records, with the UK being one of the most densely recorded countries worldwide (Powney & Isaac, 2015). Odonata, in particular, is perceived as a charismatic taxon with a vast repository of species distribution data and well-documented ecologies (Stoks & Córdoba-Aguilar, 2012). It has been recognised as a potential model taxon for investigating responses to climate change (Hassall, 2015), as it is particularly sensitive to environmental change and frequently employed as biological indicators for aquatic, and increasingly terrestrial, environments (Miguel et al., 2017). Research has already indicated that several European species' ranges are shifting or expanding northwards in correlation with recent warming (Olsen et al., 2022). Odonates have been shown to be on the move with climate change in the UK (Pettorelli et al., 2019), with arrivals of new species also documented (Cranston et al., 2023).

Based on the state of current knowledge on odonates, and the known variability in life histories in this taxon, we do not expect all odonates to exhibit niche conservatism over time (H1), with key differences in response to climate change expected between dragonflies and damselflies (Powney & Isaac, 2015). Species that show greater niche flexibility (i.e., higher level of change in their climate niche over time) could be expected to expand their distribution to new geographic areas whilst maintaining occupancy in areas experiencing changes in climatic conditions. As such, we also expect a positive correlation between niche flexibility, changes in overall range size and level of range expansion (Ralston et al., 2017; Di Cecco & Hurlbert, 2022; Tirozzi et al., 2022; H2).

6.3. Methodology

6.3.1. Species occurrence data

Species occurrence records for odonates in Great Britain between 1961 and 2020 were downloaded from both the Global Biodiversity Information Facility (GBIF, 2023) and the National Biodiversity Network Atlas (British Dragonfly Society Recording Scheme, 2023; NBN Trust, 2023). Occurrence records were prepared using the function clean_coordinates within the R package ‘CoordinateCleaner’ v.2.0–20 (Zizka et al., 2021); duplicate observations and observations with latitude and longitude values set to zero were removed. Coordinates matching country centroids and biodiversity institutions were also removed using the reference data within the package, as these are likely incorrect records or records of captured individuals.

6.3.2. Climate data

Monthly minimum and maximum temperature as well as monthly precipitation were accessed from the Met Office at a 1-km resolution (Hollis et al., 2019), covering the periods 1961-1980 (T1) and 2001-2020 (T2). This data was used to generate a series of 19 bioclimate variables using the biovars function in the R package ‘dismo’ v.1.3–14 (Hijmans et al., 2021); the variables were calculated for each year and then averaged for each time period. These 19 climate variables were chosen to be biologically meaningful and informative for Odonatan distributions (Collins et al., 2017; Abbott et al., 2022), capturing information on annual trends, seasonality and limiting environmental factors.

Despite their potential importance in shaping species distribution (Huang et al., 2017), climate extremes have received little attention when assessing the effects of climate change on biodiversity (Garcia et al., 2014). To address this, we also considered six indices that capture the intensity and frequency of climate extremes, based on the work from the Expert Team on Climate Change Detection and Indices (ETCCDI). These indices were calculated from daily temperature and precipitation data accessed from the Met Office (Hollis et al., 2019), and included: the maximum of daily maximum temperature (TXx); the minimum of daily minimum temperature (TNn); the number of frost days (FD, daily minimum temperature below 0°C); the number of summer days (SU, daily maximum temperature above 25°C); the number of consecutive dry days (CDD, precipitation below 1 mm) and the maximum 5 days precipitation (Rx5day, maximum precipitation in five consecutive days). As with bioclimate variables, each index was calculated on an annual basis and then averaged for each time period. Calculations were completed using R package 'ClimInd' v.0.1–3 (Reig-Garcia et al., 2021).

6.3.3. Assessing distribution and occupancy

The downloaded Odonatan occurrence data primarily consists of opportunistically sampled presence-only data whereby sampling effort and probability of detection are not consistent over a given area or across time. Left unaccounted for, such sampling bias can severely distort results when mapping species habitat requirements or investigating species trends (Bird et al., 2014; Johnston et al., 2021). One way to overcome this unevenness of recording – termed by Prendergast et al. (1993) 'the recorder effort problem' – is to model the data collection process (Isaac et al., 2014).

We employed the Frescalo method (FREquency SCAling Local; Hill, 2012) to correct for these data biases and to derive information on odonates distribution in Great Britain, using the Frescalo function within the R package 'sparta' v.0.2.19 (August et al., 2015). In brief, Frescalo relates records of species to a suite of local benchmark species of other odonates that vary between neighbourhoods based on the similarity between record sites. Using this method, the number of species expected to be recorded in each British National Grid hectad (10 km^2 British National Grid Squares) can be estimated, based on the local benchmark species within the neighbourhood and the level of similarity in landcover between sites based on the 2015 UK Land Cover Map data accessed through the R package. Sampling effort is calculated for each British National Grid hectad. By comparing the number of observed species for each period to the value predicted after accounting for variation in recording effort, a subset of adequately sampled hectads can be identified. These adequately sampled hectads are defined as those which reported at least 25% expected species at both T1 (1961-1980) and T2 (2001-2020), following previous research (Hordley et al., 2023). For further details, see Hill (2012).

For the purpose of this study, local benchmark species are defined as the 27% most frequent Odonatan species in a given neighbourhood, in accordance with previous research (Hill, 2012; Eichenberg et al., 2021). Time factors (TFactors) were generated to assess the reporting rate of a focal species at a given time relative to these local benchmark species. As it is advised that benchmark species should be those whose occupancy is relatively stable over time, a re-analysis was conducted, which excluded the 5% most strongly increasing or decreasing species as potential benchmark species (Hill, 2012).

Changes in occupancy, defined as changes in the number of hectads where a given species is expected to be present (Hill, 2012), can be analysed by pooling information over a series of short or long time periods. Here, we compared the outcomes of both approaches. In one approach, TFactors were extracted for each 5 years time periods throughout the duration of the study period (1961-2020) and, from these, occupancy trends were extracted through linear modelling of TFactors. T-tests were applied to determine the significance of any trend in occupancy. In the other, occurrences were pooled into two time periods (T1 and T2) and associated TFactors for these two periods were calculated, with Z-tests applied to determine significant differences. Choosing T1 and T2 as time periods meant that results were directly comparable with those from the distribution and climatic niche analysis. Species were identified as displaying significant trends in occupancy if either the 5 years approach or the two time periods approach yielded significant results.

6.3.4. Geographic distribution change

Species geographic shift was analysed in terms of the change in the geographic centre of species' distribution between periods T1 and T2 – a simplistic measure that nonetheless allows for a multidirectional assessment. The mean latitude and longitude for each time period were extracted using QGIS v.3.22.3 from species presence observations at 1-km British National Grid resolution, including only those occurrences within the previously determined adequately sampled hectads. The distance and direction between species centre at T1 and T2 were then determined using R package 'geosphere' v.1.5–18 with the functions distVincentyEllipsoid and bearing, respectively (Hijmans, 2022). This method gives an approximation of species distributional change throughout Great Britain and provides an indication of the distance and rate with which such changes can occur over a 40 years period that can be compared across species. Altitude was derived from the OS Terrain 50 digital terrain model available on the OS OpenData Downloads platform (https://osdatahub.os.uk/downloads/open). Altitude was extracted for each of the species' observations using QGIS and the difference in mean altitude between periods was computed for each species.

Average distances between the central position at T1 and T2, as well as change in altitude, were compared between damselflies and dragonflies using a Mann-Whitney U test. The direction of shift for each species was determined and analysed through a circular statistics approach conducted using the R package 'CircStats' v.0.2–6 (Lund & Agostinelli, 2018). A Watson two-sample test of homogeneity was calculated to investigate if there was any difference in the direction of species shifts between dragonflies and damselflies; this test is a non-parametric rank-based test that has been recommended for use by Landler et al. (2021) for comparing two samples of circular biological data.

In addition, a Rayleigh test of uniformity was applied to determine whether species movements followed a northwards direction (i.e., with an alternative hypothesis specified at mean = 0°) and whether dragonflies or damselflies exhibit uniform distributions. The functions circ.mean, circ.disp, r.test, v0.test and watson.two within ‘CircStats’ were employed for this analysis.

6.3.5. Climatic niche change

Species' niche change between the periods considered was determined using an ordination approach, which has been shown to quantify niche overlap more accurately than reciprocal ecological niche models when investigated using virtual species (Guisan et al., 2014). Based on direct species observations, the difference in the climatological attributes of the sites where each species was recorded in each period was compared across the available environmental space (i.e., the climatological conditions throughout the study area across both time periods), as opposed to the geographic space.

To reduce the dimensionality of the available climatic data, a Principal Component Analysis (PCA) was conducted for the 15 temperature and 10 precipitation variables, calibrated with data for the entire study area across both periods. The first two principal components were retained from the PCA, capturing 77.2% of the variability across all climate data. PC1 accounted for 58.7% of the variance and PC2 for 18.5%. For PC1, the highest positive correlation was with precipitation of the driest month (Bio14) and quarter (Bio17) and the highest negative correlation was with Bio5 (maximum temperature of the warmest month) and Bio10 (mean temperature of warmest quarter). This component can thus be understood as contrasting wet conditions from the warmest ones. The second principal component integrates information relative to the coldest conditions within a year with major contributions from seven temperature variables, with strong positive correlations with TNN (minimum of daily minimum temperature), Bio6 (minimum temperature of coldest month) and Bio11 (mean temperature of the coldest quarter) and a strong negative correlation with FD (number of frost days). Further details are provided in section 4.6 (table 4.2; figure 4.3 and figure 4.4).

PCA scores for the entire study area and each species occurrence during both time periods were subsequently projected into a grid of 1000 × 1000 cells bounded by the maximum and minimum PCA component scores across both periods – that is, the gridded climate space. This methodology – calibrating the PCA with climate data from T1 and T2 combined and subsequently calculating species' PCA scores – means that the differences in position along the principal components discriminate differences between the T1 and T2 climate space, following previous temporal analyses (Ralston et al., 2017; Sillero et al., 2022; Tirozzi et al., 2022). A smooth kernel density function was applied to estimate the density of species occurrences in each cell of the climate space, correcting for the prevalence of the environments within the species range, as an additional measure to account for sampling bias and to ensure the results remain independent of the resolution of the climate grid chosen (Broennimann et al., 2011).

The niche analysis was conducted in R with the package 'ecospat' v.3.5.1 (Broennimann et al., 2023). To determine niche change over time, Schoener's D index of niche overlap was calculated, chosen for its simplicity and long history of use (Warren et al., 2008). Schoener's D, within the context of this investigation, provides a measurement of the overlap between the climatic niche experienced by each species at T1 and T2 ranging from 0 (no overlap) to 1 (complete overlap). This index is calculated by quantifying, for each grid cell in the climate space, the difference in smoothed kernel densities of species occurrence between periods. The niche comparison between periods was thereupon decomposed into three components to provide additional information about the drivers of niche change (Tirozzi et al., 2022): (i) niche expansion, the proportion of species' niche present at T2 only; (ii) niche contraction (often termed niche unfilling), the proportion of species' niche no longer present at T2 and (iii) niche stability, the proportion of niche present at both time periods. Statistical tests for niche similarity between periods were used to determine whether observed climatic niches at T2 were statistically similar to the observed climatic niches at T1 using the ecospat.niche.similarity function within ecospat. The hypothesis for niche conservatism was tested by comparing observed climatic niches at T1 with random simulations of species' niches at T2 within the available climate space with the same kernel density distribution as T1, following the approach set out by Broennimann et al. (2011). By repeating these simulations 100 times, a null distribution of species overlap values was generated to determine if observed species niche overlaps are more statistically similar (niche conservatism) or not (niche divergence) than random, based on a significance threshold of 0.05. To determine whether there was any significant difference between dragonflies and damselflies in the proportion of species whose niche was conserved over time, a two-proportion Z-test was conducted in R.

The overall approach – calculating niche overlap, determining niche dynamic indices and testing for niche conservatism through niche similarity testing – was first conducted including the full range of climatic conditions across both time periods. This was to provide a complete picture of the climatic niche changes that have occurred throughout the entirety of Great Britain, including novel climatic conditions and changes between the first and the second time periods. This analysis was then repeated at the intersection of climates between the two time periods, termed the analogue climate analysis, to evaluate species niche dynamics considering only the climatic variability available during both periods thereby excluding niche shifts resulting from variation in the climatic space accessible to species (Tirozzi et al., 2022); subsequent results refer to this method unless specified otherwise.

6.3.6. Geographic versus climatic niche shifts

Kendall correlation analyses were conducted to look for any significant relationship between geographic and climatic niche shifts; this type of analysis was deemed an appropriate non-parametric correlation test able to accommodate ties (Tirozzi et al., 2022). Correlation tests were run between species overlap and geographic distribution change, including distance of centroid shift and change in latitude and longitude of these distribution centres. Kendall correlation tests were also employed to determine whether there was any significant correlation between species occupancy trends and niche dynamics, including Schoener's D index of climate niche overlap, expansion and contraction.

6.4. Results

Of a total of 37 species of odonates considered, 29 species (78.4%) displayed significant changes in occupancy between 1961-1980 (T1) and 2001-2020 (T2). Sixteen (43.2%) of them showed significant increases in the area they occupy in Great Britain, whilst 13 species (35.1%) showed significant decreases in this parameter. Species' centre of distribution shifted between 3.4 km (*Coenagrion hastulatum* Charpentier, 1825) and 230.8 km (*Leucorrhinia dubia* Vander Linden, 1825) over the study period, equating to a rate of between 0.09 km and 5.8 km a year. Overall species shifts were in an approximately northwards direction (circular mean of 10.5 ± 25.1°), ranging primarily from an approximately northwest direction (300.4°, *Somatochlora arctica* Zetterstedt, 1840) to a northeast direction (76.7°, *Lestes dryas* Kirby, 1890), with only one species shifting southwards (220.2°, *Aeshna isosceles* Müller, 1767; table 6.1; figure 6.1). Rayleigh's test for uniformity supported the presumption that species do have a tendency to shift in a northwards direction ($p < .001$). Both dragonflies and damselflies occurred at higher altitudes on average at T2 than at T1 (with a mean change 3.36 ± 22.2 m and 2.64 ± 17.5 m between periods, respectively), but with high variability between species. Eight damselflies and seven dragonflies had a lower mean altitude at T2 than T1 (40.5% total species).

Table 6.1. Odonates geographic distribution change including annual occupancy trends (ordered from greatest increase in occupancy to greatest decrease, with 0 indicating no significant change detected) and species shift in distribution centre in terms of annual distance and direction of shift.

Species	**Suborder**	**Annual Occupancy Change (%)**	**Annual Centroid Shift: Distance (km)**	**Centroid Shift: Direction**
Anax imperator	Anisoptera	0.75	2.20	22° (N)
Sympetrum sanguineum	Anisoptera	0.60	1.90	18° (N)
Aeshna mixta	Anisoptera	0.53	1.94	353° (N)
Aeshna isoceles	Anisoptera	0.42	0.67	220° (SW)
Libellula depressa	Anisoptera	0.41	2.20	26° (NE)
Orthetrum cancellatum	Anisoptera	0.33	1.70	2° (N)
Brachytron pratense	Anisoptera	0.32	1.97	34° (NE)
Erythromma najas	Zygoptera	0.31	1.06	17° (N)
Libellula quadrimaculata	Anisoptera	0.30	2.16	23° (NE)
Libellula fulva	Anisoptera	0.20	1.38	12° (N)
Somatochlora arctica	Anisoptera	0.19	0.29	300° (NW)
Orthetrum coerulescens	Anisoptera	0.18	2.05	358° (N)
Lestes dryas	Zygoptera	0.15	1.77	77° (E)
Ischnura pumilio	Zygoptera	0.15	0.93	352° (N)
Platycnemis pennipes	Zygoptera	0.14	1.17	27° (NE)
Calopteryx splendens	Zygoptera	0.12	1.62	26° (NE)
Aeshna caerulea	Anisoptera	0.00	1.03	342° (N)
Aeshna cyanea	Anisoptera	0.00	1.30	355° (N)
Aeshna juncea	Anisoptera	0.00	3.36	353° (N)
Calopteryx virgo	Zygoptera	0.00	1.70	56° (NE)
Coenagrion hastulatum	Zygoptera	0.00	0.09	326° (NW)
Coenagrion mercurial	Zygoptera	0.00	0.71	323° (NW)
Gomphus vulgatissimus	Anisoptera	0.00	1.41	315° (NW)
Somatochlora metallica	Anisoptera	0.00	2.84	337° (NW)
Sympetrum danae	Anisoptera	-0.15	3.52	349° (N)
Ceriagrion tenellum	Zygoptera	-0.24	0.80	71° (E)
Cordulegaster boltonii	Anisoptera	-0.24	1.57	6° (N)
Cordulia aenea	Anisoptera	-0.25	0.90	352° (N)

Lestes sponsa	Zygoptera	-0.29	1.93	11° (N)
Coenagrion pulchellum	Zygoptera	-0.33	1.38	48° (NE)
Sympetrum striolatum	Anisoptera	-0.41	1.75	25° (NE)
Leucorrhinia dubia	Anisoptera	-0.41	5.77	344° (N)
Pyrrhosoma nymphula	Zygoptera	-0.48	1.40	50° (NE)
Enallagma cyathigerum	Zygoptera	-0.51	0.86	71° (E)
Aeshna grandis	Anisoptera	-0.65	0.72	13° (N)
Coenagrion puella	Zygoptera	-0.78	1.55	28° (NE)
Ischnura elegans	Zygoptera	-1.11	0.98	33° (NE)

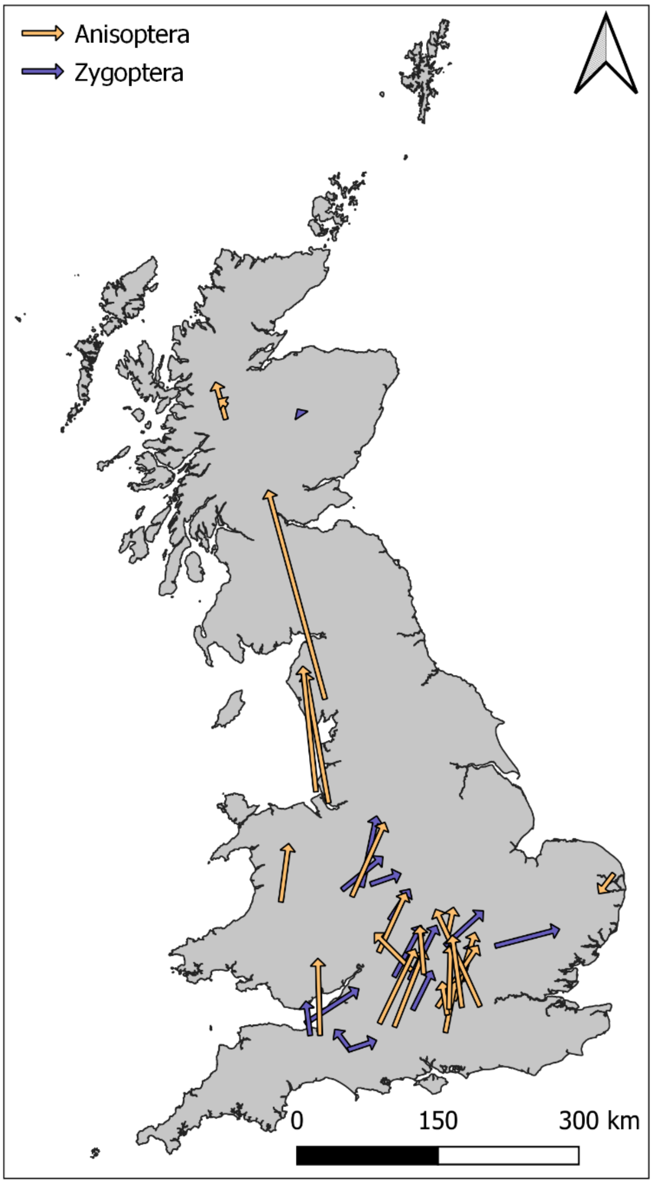


Figure 6.1. Shifts in dragonflies (Anisoptera) and damselflies (Zygoptera) distribution centre between periods T1 (1961-1980) and T2 (2001-2020).

Our results support the hypothesis that not all odonates in Britain exhibited niche conservatism over the period considered (H1). Sixteen species (43.2%) displayed statistically more similar climate niches between T1 and T2 than expected by chance, indicating niche conservatism; but the remaining 21 species (56.8%) showed signs of niche divergence. When considering the full range of climatic variability across Great Britain, 20 species exhibited niche conservatism (54.1%). Schoener's D index of climate niche overlap between time periods varied from 0.00 (*A. isoceles*) to 0.43 (*Libellula depressa* Linnaeus, 1758). Four species of dragonfly (*A. isoceles*, *Gomphus vulgatissimus* Linnaeus, 1758, *Libellula fulva* Müller, 1764, *Somatochlora metallica* Vander Linden, 1825) and two species of damselfly (*C. hastulatum, L. dryas*) had expansion values over 80%. Niche contraction was typically low, with 21 species (56.8%) having less than 20% niche contraction and 26 species (70.3%) having less than 30% contraction. The species with the highest proportion of niche contraction were *A. isoceles* (100%), *L. dryas* (96.6%), and *C. hastulatum* (66.9%; figure 6.2).

Figure 6.2. Niche dynamics across dragonflies (Anisoptera) and damselflies (Zygoptera) considering all climatic variability and analogue climate between periods.

As expected, comparing dragonflies and damselflies unveiled several key differences between the two. For example, the 10 species with the greatest distribution shift in terms of distance between distribution centres were all dragonfly species. Dragonflies shifted further than damselflies on average (Mann-Whitney U test, $p = 0.02$) with annual mean shifts of 1.9 ± 1.18 km and 1.2 ± 0.49 km, respectively. The distributions of the direction of shift of both dragonflies and damselflies were non-random (Rayleigh test, $p < .001$), but these were not consistent between both groups of species (Watson two-sample test, $p < .01$): dragonflies shifted in a mean northwards direction (357.4°, SD = 22.4°) whilst damselflies shifted, on average, a mean northeast direction (30.3°, SD = 23.4°). However, there was no statistical difference in change of altitude (Mann-Whitney U test, $p = .55$) or the proportion of species with divergent niches ($z < 0.001$, $p = 1$) between dragonflies and damselflies.

As expected under H2, an overall significant increase in occupancy was correlated with an overall decrease in climatic niche overlap between T1 and T2 for damselflies (Kendall correlation, $\tau = -0.54$, $p = .005$). For that group of species, we moreover found a positive correlation between occupancy trends and climatic niche expansion ($\tau = 0.52$, $p = .007$), which was expected, but also a positive correlation between occupancy trends and contraction ($\tau = 0.43$, $p = .03$), which was less expected. For dragonflies, however, we found no significant correlation between occupancy trends and niche overlap ($\tau = 0.07$, $p = .67$), expansion ($\tau = 0.12$, $p = .44$) or contraction ($\tau = 0.004$, $p = .98$; figure 6.3).

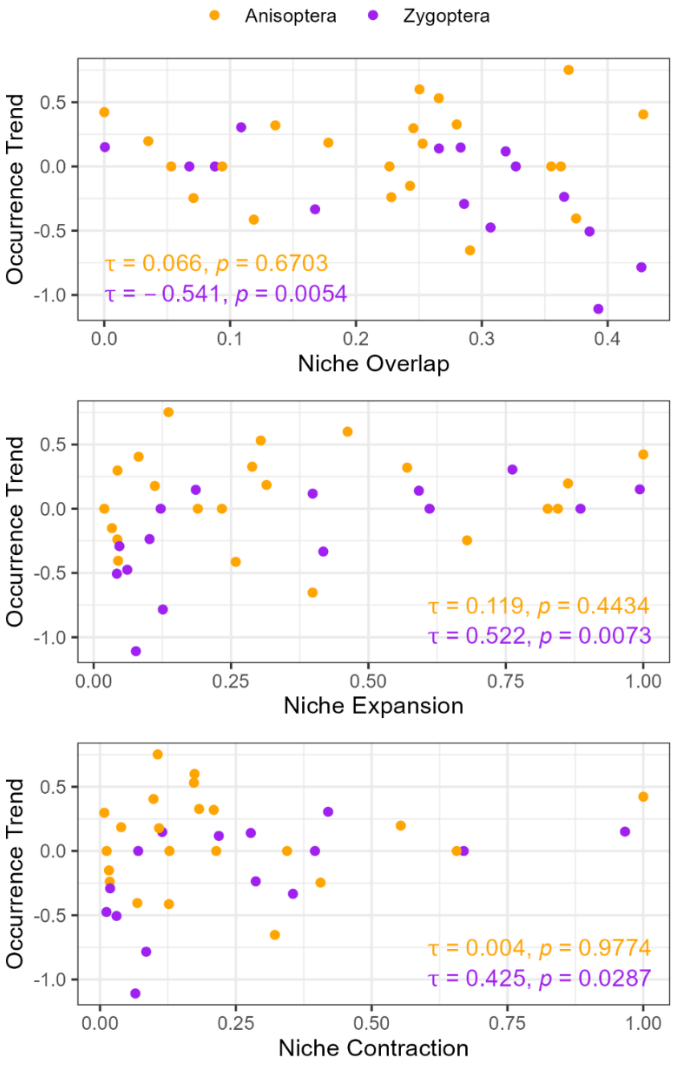


Figure 6.3. Kendall correlation between species occurrence trends and species climatic niche dynamics (namely Schoener's D overlap index on the top, niche expansion in the middle and niche contraction on the bottom).

For dragonflies, a Kendall correlation test revealed a significant positive correlation between niche overlap and change in longitude (τ = 0.32, p = .04) but no significant correlation with change in latitude (τ = 0.07, p = .63), change in altitude (τ = −0.22, p = .16), or distance between distribution centres (τ = 0.06, p = .71). Whereas considering the full range of climate values and novel climates at T2 revealed a significant positive correlation between niche overlap and change in latitude (τ = 0.33, p = .03). Dragonflies with the greatest shifts towards higher latitudes and longitudes thus had the largest consistency in climatic niche between T1 and T2. For damselflies, no Kendall correlation was detected between occupancy trends and distance between distribution centres (τ = 0.10, p = .59), nor change in latitude (τ = 0.24, p = .46), longitude (τ = 0.14, p = .22) or altitude (τ = −0.31, p = .11).

6.5. Discussion

This study demonstrates for the first time how recent changes in climatic conditions triggered alternative responses by odonates in Great Britain, with some species shifting their geographic distribution to track preferred climatic conditions in space, and others shifting their realised climatic niche. Contradictory to previous assumptions that species niches remain consistent over time, we found evidence that several odonates have not retained statistically similar realised climatic niches in Great Britain during a period of recent climate change. For damselflies, species with greater differences in their realised climatic niche between periods correlated positively with an increase in species' occupancy, suggesting that niche flexibility underpinned the dispersion of damselflies into new geographic areas. For dragonflies, species distributional changes towards higher latitudes and longitudes enabled those species to maintain a greater climatic niche overlap between periods, providing evidence for climate tracking. Overall, this research provides a framework for assessing multidirectional distribution shifts alongside climate niche dynamics.

Species distribution shifts with climate change have been widely documented for a range of species, including odonates. Several European species have been reported to shift northwards with recent climatic warming (Olsen et al., 2022), with odonates being among the taxa showing the strongest polewards range expansions (Bybee et al., 2016). Our centroid change analyses showed that species shifted up to 5.8 km annually, which is comparable to previous studies (Hickling et al., 2005). Species exhibiting niche fidelity between periods – with statistically similar climatic niches between periods and a large proportion of niche overlap – might be concluded to be tracking climatic change across the landscape, if there is any evidence of a distributional change. Yet it might not always be the case that the species with the greatest shift or the greatest change in latitude and longitude have the most consistent climatic niche as species' specific climatic tolerances might not always necessitate large distributional change. Species on the move with climate change might therefore be classified as any species with a statistically similar niche during a period of climatic change that has had a significant positive trend over the same period or whose distribution centre has altered to a certain degree.

Dragonflies shifted greater distances than damselflies on average, likely because they are typically larger, faster and more robust than damselflies (Cancellario et al., 2022; Olsen et al., 2022). Dispersal capacity has been previously found to constrain species ability to geographically track climate change (Thomas et al., 2010; Littlefield et al., 2019) and might similarly substantiate an important consideration for climatic niche change analyses. Whilst species with greater dispersal abilities might have greater capacity to shift into newly available climatic space, those with lower dispersal ability, such as damselflies, may not be able to shift geographically in pace with climatic change and may instead be left behind in unsuitable climate. Considering these geographic and climatic niche changes alongside change in species occupancy allows an examination of which species are experiencing the greatest increase or decline in their distribution in Great Britain and an assessment of the relative success of their varying responses. Overall, our results imply that the best dispersers may be more able to maintain consistent climatic niches over time, whilst species with narrow thermal tolerances and low dispersal ability may be more likely to display climatic niche flexibility.

This study refers to species realised climatic niche based on the climate variables throughout species geographic distribution within Great Britain, and so all values refer to species' niche dynamics on a national scale and are not informative of species' climatic niche within their wider distribution. As such, species which have not exhibited a statistically similar niche between periods within Great Britain, could have exhibited niche conservatism overall had their entire geographic distribution been considered beyond this national boundary. To aid the interpretation of these national findings, species realised niche conservatism was computed considering an analogue climate analysis – including only the climatic variability available during both time periods examined. Thereby, even though species might occupy a broader climatic range outside of Great Britain, any changes of species' niche within a given limited set of climatic conditions theoretically available to species can be established. Furthermore, our findings indicate differences in species niche dynamics within a geographically isolated landmass and these results are relevant to national-level studies, a scale that distribution models and climate change assessments are often applied.

Our results may have important ecological and conservation consequences as they cast doubt on typical assumptions underpinning species distribution modelling and other investigations where species climate niche is thought to remain broadly stable over time at national scales. They also suggest that efforts to theorise future species distributions based on vectors of climate change might prove to be an inappropriate basis for allocating conservation efforts, without prior consideration as to whether species climatic niche can change over time. Such findings are in accordance with other investigations that have found shifts in the realised climatic niches of plants, birds, reptiles, amphibians and mammals during recent climate change (Wolf et al., 2016; Enriquez-Urzelai et al., 2019; Sillero, 2021; Sillero et al., 2022;). Broennimann et al. (2011) advocated for avoiding the projection of niche models outside the calibration area unless niche overlap is high (Schoener's D ≈1) and tests for niche equivalency could not be rejected; here even those 16 species which exhibited statistically similar niches over time had low niche overlap (maximum 0.43) and would not meet such requirements to adequately project future distributions within Great Britain under continued climate changes. The application of SDMs for predicting Odonatan distributions based on similar bioclimatic data using national scale occurrence information such as from the British Dragonfly Society, should therefore be implemented with care; whether species are exhibiting niche flexibility under climatic change, or whether national distribution data is inadequate in defining species' climatic niche, projections may not be accurate for Odonata and potentially other species. SDMs based on climatic and environmental data calibrated with national occurrence data within Great Britain have been previously employed for Odonates (Geary & Von Hardenberg, 2020), other invertebrates (Giannini et al., 2012; Oliver et al., 2012; Bourhis et al., 2023), as well as plants, birds, reptiles, amphibians and mammals (Oliver et al., 2012; Rapacciuolo et al., 2014; Buxton et al., 2021; Rodríguez-Roy et al., 2023); including species of conservation concern (the grey long-eared bat *Plecotus austraicus*; Razgour et al., 2011) and invasive species (Reeve's muntjac deer *Muntiacus reevesi*; Freeman et al., 2022). Further temporal analyses of climatic niche changes within broader geographic and temporal scales would be advantageous to further evaluate niche conservatism.

It should also be emphasised that differences here refer to species realised climatic niche that is restricted by historical and biotic factors and is a subset of the broader environmental conditions that species could potentially occupy – the fundamental niche (Jezkova & Wiens, 2016). Isolating these aspects would provide a greater understanding of species response to climate change; however, this has proved problematic to date (Guisan et al., 2014). The reported niche shifts experienced by some species may indicate a change in fundamental niches resulting from evolutionary adaptation to climate change. But these shifts could also be due to species being unable to keep pace geographically to remain within their optimal climate conditions. Alternatively, it may also be the case that species displaying changes in their climate niches over the period considered are simply not limited by these climatic factors within their Great British distribution. Further research into the specific climatic tolerance of the species considered in this study would enable a greater understanding of the mechanisms underpinning our findings.

Although Odonatan suborders demonstrated significantly different responses to climate change, we found a noticeable level of variability between species in each suborder, in line with previous reports on species redistribution (Hill et al., 2002; Angert et al., 2011). For example, niche conservatism and niche divergence were both found among dragonfly and damselfly species, corroborating trends found in other temporal assessments of realised climatic niches (Tirozzi et al., 2022). This level of interspecific variation, and the lack of congruence in responses across phylogenetically similar groups, calls for more research across a wider set of taxa on the factors promoting niche conservatism or niche divergence in response to climate change. Such knowledge will be key to inform wildlife management as our climate continues to alter.

7. Insight from invasion biology for assessing the impacts of climate-tracking Odonata and Hymenoptera

7.1. Abstract

The geographic movement by species to track their optimal climate conditions is a widely reported phenomenon, however, the resulting environmental impacts are insufficiently understood. Furthermore, there are no structures currently in place for the evaluation or grading of potential ecological impacts. It has been theorised that frameworks within invasion biology provide the means to assess the potential impact of climate-tracking species, however, this has not been practically tested. Here invasion frameworks are employed for the first time for native species which are shifting due to climate change, using Odonata (dragonflies and damselflies) and Hymenoptera (wasps, bees, ants and sawflies) species in Great Britain as a case study. Potential impacts to other native species are assessed through a literature review and the level of impact graded following the invasion protocols. A spatial impact assessment is conducted by mapping species changing distributions under a future climate scenario. 97 individual impacts were reported across six Odonata and eleven Hymenoptera species through a range of different mechanisms, both positive and negative. 99 native species were reported to be impacted by the climate-tracking Hymenoptera and 18 by the Odonata; these impacted species were largely other Insecta (67.5%, N = 79). A substantial quantity of the reported impacts was of Minimal Concern (MC) or Minimal (ML+) positive impact (48.7%, N =19), however, two impacts (5.1%) resulted in an increase in occupancy of another native species, nine (23.1%) caused an increase or decrease in the population size of another native species, and a further nine (23.1%) impacted another native species' performance. This research demonstrates the potential applicability of an invasion framework for species undergoing shifts due to climate change in providing a possible method for examining the ecological impacts for other native species.

7.2. Introduction

When faced with rapid changes in long-term climatic conditions, one mechanism by which species can survive and persist, is to geographically shift to track their optimum climate niche (Scheffers et al., 2016). As examples of such climate-driven range shifts proliferate, scientists and conservationists alike are becoming aware of the vast implications of such movements for ecosystems and the societies they support. It is crucial for the future management of biodiversity for structures to be put in place for the detection of species movements due to climate change, as well the full evaluation of potential environmental and socio-economic impacts of these shifts – both positive and negative; yet there is no formal framework in place for determining or reporting the impact of such species on recipient environments.

Within invasion biology, several structures already exist to evaluate the consequence of species arriving in new locations that could potentially be adjusted to focus on the assessment of species that are shifting due to climate change (Wallingford et al., 2020). One approach for invasive species, is the use of an impact categorisation and scoring system, first developed by Nentwig et al. (2010). Based on such concepts and the widely accepted Red List of Threatened Species, Blackburn et al. (2014) developed the Environmental Impact Classification of Alien Taxa (EICAT) approach whereby the magnitude of impact from Minimal Concern (MC) to Massive (MV) is classified according to semi-quantitative scenarios under specific identified mechanisms of impact such as predation, competition, and parasitism. The EICAT system was later adapted by Vimercati et al. (2022) to form the EICAT+ to categorise the positive impacts of non-native species, with additional mechanisms through which alien species can benefit native taxa, whilst the EICAT focusses on negative impacts. As species that have tracked their optimum climate into new locations can positively and negatively impact recipient environments through the same mechanisms as invasive species, theoretically such assessment methodologies could be employed for climate-tracking species.

Although such parallels have been recognised within literature (Pettorelli et al., 2019; Wallingford et al., 2020), the practical application of such insights from invasion biology for species redistribution remains under-investigated. Furthermore, there are no examples of invasion frameworks having been directly employed for climate-tracking species, despite a concern of potentially severe and wide-reading anticipated impacts of species redistribution for ecosystems (Twiname et al., 2020; Wallingford et al., 2020), and that such methods within invasion biology have been adopted by the International Union for Conservation of Nature (IUCN) and widely applied. This research provides a novel insight by employing the EICAT and EICAT+ frameworks to assess negative and positive impacts, respectively.

These methods are demonstrated by employing a case study comparing Great Britain’s Odonata (dragonflies and damselflies) and Hymenoptera (sawflies, wasps, bees, and ants) species. Great Britain offers a vast quantity of biological records, with the UK being one of the most densely recorded countries worldwide (Powney & Isaac, 2015). Odonata have been recognised as a potential model taxon for investigating responses to climate change (Hassall, 2015), being particularly sensitive to environmental change (Miguel et al., 2017). Odonates are of ecological significance being important predators in both terrestrial and aquatic environments, with Odonate larvae being top predators in several fish-free water bodies (May, 2019) and having a wide diversity of prey taxa (Sentis et al., 2022). Odonates likewise provide prey for both terrestrial and aquatic species including spiders, birds, and fish (Sentis et al., 2022). Furthermore, both reproductive and aggressive interference are frequent amongst species (Grether et al., 2022). Hymenoptera are likewise sensitive to alterations in temperature and precipitation and several species directly depend on the availability of flowering plants (Ali et al., 2023). The Hymenoptera order contains numerous invasive species, with impacts such as parasitism, competition, and hybridisation reported (Rasplus et al., 2010). Several Hymenopteran species also provide vital ecosystem services being effective agents for pest regulation and having an important role for pollination (Rasplus et al., 2010; Queffelec, et al., 2021). Therefore, both Odonata and Hymenoptera represent ecologically interesting groups to investigate the potential environmental influences of climate-tracking species. Moreover, research has already indicated that several Odonatan and Hymenopteran species' ranges have shifted with recent warming, including in the UK (Burton, 2003; Pettorelli et al., 2019; Olsen et al., 2022).

Hymenoptera are predicted to be a more problematic group, with a greater severity of negative impacts. These impacts are theorised to occur through EICAT mechanisms reported for invasive Hymenoptera worldwide on the Global Invasive Species Database (GISD; http://www.issg.org/database) – i.e. competition, predation, transmission of disease, parasitism, and poisoning/ toxicity. Not being invasive, Odonata are expected to have only minor deleterious impacts under EICAT if any, with potential mechanisms being predation, as well as competition with other Odonates. It is hypothesised that species of both groups will have positive impacts through the provision of trophic resources to other native taxa, for Hymenoptera dispersal facilitation is also predicted to be important as many species are important pollinators.

Species arrivals under climate change will depend on their current range within Great Britain, and their climatic restrictions. The selected species which have maintained consistent climate conditions whilst expanding their distribution during recent climate change are theorised to largely consist of southerly distributed species with their northern range boundary within Great Britain which are expanding northwards. Therefore, species arrivals and associated negative and positive impacts under future projected climate change are theorised to be low in the south where species are already present, with the highest impacts being in the midlands and towards the north, depending on where species current range margins lie. Species might also be expected to shift westwards to areas of higher rainfall, particularly for Odonata, with the southeast largely becoming too hot and dry for our current natives.

7.3. Methodology

7.3.1. Species occurrence data

Odonata and Hymenoptera species occurrence records in Great Britain for the period, 2001-2020 were downloaded from both the Global Biodiversity Information Facility (GBIF, 2023) and the National Biodiversity Network Atlas (Biological Records Centre, 2023; British Dragonfly Society

Recording Scheme, 2023; Bumblebee Conservation Trust, 2023; BeeWalk bumblebee distributions for Great Britain 2008-2022, 2023; Bee, wasp and ant (Hymenoptera: Aculeata) records verified via iRecord, 2023; NBN Trust, 2023; Sawfly (Symphyta) records from iRecord, 2023), as well as additional data for Hymenoptera acquired directly from the Bees, Wasps and Ants Recording Society. Occurrence records were prepared using the function clean_coordinates within the R package ‘CoordinateCleaner’ v.2.0–20 (Zizka et al., 2021); duplicate observations and observations with latitude and longitude values set to zero were removed. Coordinates matching country centroids and biodiversity institutions were also removed using the reference data within the package, as these are likely incorrect records or records of captured individuals.

7.3.2. Environmental data

Climate averages for rainfall, minimum and maximum temperature for the period 1991 to 2020 was retrieved from the Met Office (Met Office et al., 2023) on a 5-km grid for the UK to provide a baseline for Species Distribution Model (SDM) projections. Climate projections from the United Kingdom Climate Projection 2018 (UKCP18) project were downloaded at a 5-km resolution from the Met Office Hadley Centre (2019) for the period 2020-2040 under a high emissions scenario, Representative Concentration Pathway 8.5 (RCP8.5). The twelve Met Office Hadley Centre climate model projections for rainfall, minimum and maximum temperature were all downloaded, and a mean calculated for each variable across these models. This data was used to generate a series of 19 bioclimate variables using the biovars function in the R package ‘dismo’ v.1.3–14 (Hijmans et al., 2021) that are biologically meaningful and informative for Odonatan and Hymenopteran distributions (Collins et al., 2017; Abbott et al., 2022), capturing information on annual trends, seasonality, and limiting environmental factors.

In addition to climatic data, information for the distribution of broadleaf forests was also obtained as three identified climate-tracking species are gall wasps whose distribution explicitly relies on the presence of oak woodland; however, data was not available at the species level. Therefore, the National Forest Inventory GB 2020 was downloaded from the Forest Commission Open Data (2023) which is a shapefile file of Great Britain’s woodland. To prepare this data on the same resolution as the climatic data, the broadleaf area was first extracted, and the percentage coverage calculated for each 5-km British National Grid using QGIS. Although not ideal, the broadleaf information might still be indicative of gall species’ potential habitat suitability and can be retained if deemed important when environmental variables are reviewed.

7.3.3. Selection of study species

To select study species within the Odonata and Hymenoptera groups, species' current distribution and realised climatic niche was compared with historic data employing methods from **chapter 6**. Species were chosen that had experienced geographic expansion during a period of recent warming (see section 4.4.1.a) whilst maintaining within a consistent climatic niche (see section 4.6.4). Such species are responding to climate change in a predictable way and likely to continue to arrive in new areas within Great Britain under continued climatic change. The study species are thus termed climate-tracking species, as they have been able to remain within their past climatic conditions as the climate has altered.

7.3.4. Projection of species' future distribution

Ensemble SDMs for each species were implemented in R using the 'biomod2' package (Thuiller et al., 2021). Different model algorithms were compared in terms of habitat suitability outputs and model performance metrics and six modelling techniques selected, each with the default biomod2 settings; these included three machine-learning methods, generalized boosting model, random forest, and maximum entropy; two regression methods, generalized linear model, and multiple adaptive regression splines; and finally, a recursive partitioning method, classification tree analysis. For each species, 1,000 pseudo-absence points were randomly selected from the background data at least 5.5 km away from any observed occurrences to reduce the potential of selecting pseudo-absence points within the same niche as presences. This quantity was chosen due to the broad distribution of some species within the study area limited the availability of background data, whilst being approximately proportional to the less numerous occurrence datasets and therefore broadly appropriate across species (Barbet-Massin et al., 2012).

Sampling bias of opportunistically sampled data can severely distort results of SDMs that rely on the assumption that sampling bias and probability of detection are approximately even over a given area (Bird et at., 2014; Johnson et al., 2021). To account for this sampling bias when applying modelling algorithms, a 'bias covariate correction' method was employed (Warton et al., 2013; Chauvier et al., 2021), where proxies for sampling effect are used to correct for bias. Several potential sampling effort predictors were reviewed – percentage cover of built-up areas and gardens, distance to the nearest road, population density, and visible night light. Percentage cover of built-up areas was extracted from the UK Centre for Ecology and Hydrology 2021 Land Cover Map at a 1-km resolution (Marston et al., 2022). The OS Open Roads (2023) vector dataset was downloaded for the UK from EDINA Digimap, and the distance from each 5-km grid cell in the study area to the nearest road was calculated in QGIS. Residential gridded population data was also acquired from Digimap, which is based on the 2011

Census and 2015 Land Cover Map (Reis et al., 2017). Annual composites of visible night light version 2 for the years 2016 to 2020 were downloaded for the Earth Observation Group (Elvidge et al., 2021) and averaged across these years. Predictor covariation was assessed by calculating the Pearson's correlation coefficient between each pair of variables and a preliminary set of models was run to check for variable importance to guide predictor selection. Importance was determined by computing the Pearson's correlation between predictions made with a given variable and with the variable replaced with a randomized input, with variable importance averaged from five permutations. Where sampling effort variables were important ($1 - r > 0.05$, where r is the Pearson's correlation coefficient), they were retained in the final model, and when variables were correlated (Pearson's correlation > 0.7), only the most important variable was included. This resulted in the inclusion of the distance to nearest road and visible night light as the chosen recorder effort proxies. Model algorithms were trained with these variables alongside selected predictor variables, then were set to the median value across the study area for projection of current and future habitat suitability thereby compensating for the potential effect of sampling effort following the protocol of Warton et al. (2013).

Climate predictor variable selection followed the same procedure, whereby the correlation between climate variables was calculated to ensure that no correlated climate variables were included, either for the current or future climate projection. Preliminary model projections under current and future climate scenarios were also modelled for a subset of four species (the most and least densely recorded species for both taxonomic groups) including all climate variables to assess variable importance and model performance, to guide climate predictor selection. These preliminary screening steps resulted in a final set of predictors consisting of minimum temperature of the coldest month (bio6), temperature annual range (bio7), mean temperature of the warmest quarter (bio10), and precipitation of the wettest quarter (bio16), as well as percentage broadleaf woodland cover.

Each model was evaluated using a 5-fold cross-validation technique, whereby 20% of the presence and pseudo-absence data was set aside for evaluation, repeated over five model runs each with a different 20% subset set aside for evaluation. Model performance was assessed with the true skill statistic (TSS), which provides a threshold-independent measure of accuracy (Allouche et al., 2006) and is the recommended approach when attempting to maximize discrimination in SDMs (Steen et al., 2020). Several alternative metrics were also calculated to provide an overall summary of performance and potential bias. These included Area Under the receiver operating Curve (AUC); frequency bias, the ratio between observed and predicted presences; accuracy, the fraction of occurrences correct; and finally, Cohen's Kappa coefficient, a measure of model accuracy which corrects for accuracy expected to occur by chance (Allouche et al., 2006). Ensemble models were built combining all individual models with a TSS value greater than 0.4, considered to be useful (successfully able to discriminate suitable from unsuitable areas; Zhang et al., 2015; Soultan & Safi, 2017), and weighing model contribution according to TSS scores.

The current and future distribution of species was therefore projected by employing these ensemble models by training model algorithms on species' current distribution under recent climate conditions and subsequently predicting habitat suitability across Great Britain under these current climate conditions as well as under future projected climate conditions. Habitat suitability projections were converted to binary presence-absence maps by selecting a threshold maximizing the TSS score of ensemble model predictions for each species. The difference in total distribution area across Great Britain between current and future projections was calculated at the 5-km resolution in which distributions were projected. The projected arrival of species in new locations within Great Britain could then be delineated by subtracting the current modelled species presence from the future modelled presence, and likewise the predicted loss of potential habitat determined by subtracting the future modelled presence from the current projection. The total geographic area of species projected current and future distributions were extracted from binary maps in QGIS and the percentage change calculated. The total area of expansion and contraction of suitable habitat area were also determined.

7.3.5. Employing EICAT and EICAT+

The potential adverse impact of climate-tracking species was assessed by following the formal EICAT procedure outlined by the International Union for the Conservation of Nature (IUCN, 2020a; IUCN, 2020b). In brief, a systematic review is conducted to gather observations of deleterious impacts by alien taxa to native taxa which are categorised under twelve impact mechanisms. Each impact mechanism is assessed against five criteria and an EICAT category assigned as the highest level of criterion met under each impact mechanism. The impact categories range from Minimal Concern (MC) whereby no reduction in performance of native taxa is detected to massive (MV) where the alien taxa result in the local extinction of a native taxon. The potential positive impacts were assessed with reference to the EICAT+ framework developed by Vimercati et al. (2022) to complement that of EICAT. EICAT+ includes ten impact mechanisms, some with sub-mechanisms, that either directly map onto EICAT (for example a species may have a deleterious impact by predating on native taxa or a positive impact through the provision of trophic resources by being prey for a native taxa) or are an additional mechanism, such as dispersal facilitation. Impacts under these mechanisms are assigned an EICAT+ category corresponding to that of EICAT, ranging from Minimal (ML+) positive impact to Massive (MV+) positive impact. For both EICAT and EICAT+, a confidence rating is assigned, according to the potential presence of confounding effects, study design, data quality and type, spatial and temporal scale, and the coherence of evidence.

To implement the EICAT/EICAT+ protocol for the selected study species to assess potential negative/positive impacts for other native species, required some minor adjustments. The systematic literature review of potential impacts was conducted through a literature search on the Web of Science. As conducted for native species, relevant literature was gathered by employing search terms specifically targeted for each impact mechanism under EICAT and EICAT+. Search results were subsequently collated for each species and screened firstly with reference to the abstract and title and then the full text. Studies that detailed an impact by the climate-tracking species on a UK native species through one of the relevant impact mechanisms were included. For EICAT/EICAT+ impacts are required to have been reported within the same context under which they are being assessed, however, for this assessment observations reported from outside the UK were included provided they impacted a UK native species; such results were thereby assigned a confidence rating of 'low' due to the uncertainty resulting from spatial extrapolation. Each result was then assigned to an impact category following the EICAT and EICAT+ procedure, with the only deviation being for the assessment under the predation mechanism. For EICAT, predation of a native taxa is assigned to a minimum impact category of Minor (MN), even without evidence of a resulting decrease in performance of a native taxa, whereas under this assessment this limit was not employed and the assessment for predation remained consistent with the other mechanisms.

7.3.6. Spatial impact assessment

To assess the potential future impact of species on the move in Great Britain, the focus was on the positive and negative impact of species on other native taxa, when arriving in a new locality. Therefore, to map the impacts for both taxonomic groups the number of new climate-tracking species projected to arrive in each 5-km British National Grid was calculated. The impacts of each species under each recorded impact mechanism were given a score based on the impact category prescribed; positive impacts assessed under EICAT+ were given positive scores ranging from 1 for a Minimal (ML+) positive impact to 5 for a Massive (MV+) positive impact and deleterious impacts assessed under EICAT were ascribed negative scores from –1 for Minimal Concern (MC) to –5 for Massive (MV) impact. A spatial assessment of the impact of climate-tracking species could then be conducted by mapping species projected arrivals under a future climate scenario alongside the associated impact score for each impact mechanism for each species and totalling the impact score across all Odonatan and Hymenopteran species arrivals in each 5-km grid. Following this method, maps of potential species impacts were produced for both taxonomic groups, separately for positive and negative impacts, as well as for the net potential impacts resulting from both negative and positive impact mechanisms.

7.4. Results

7.4.1. Selection of study species

There was a sufficient quantity of occurrence data (>5 observations in both periods) to investigate 38 Odonata species and 512 Hymenoptera; of these 17 (44.7%) Odonata species and 268 (52.3%) Hymenoptera species were found to have increased in distribution size between 1961-1980 and 2001-2020. As anticipated, not all species exhibited a statistically similar climatic niche in Great Britain between periods, in fact only 19 (50.0%) Odonata and 41 (8.0%) Hymenoptera species exhibited niche conservatism (a more statistically similar niche then expected by chance). The selected study species were those exhibiting niche conservatism and expanding between periods, constituting 9 (23.7%) Odonata and 15 (2.9%) Hymenoptera species (table 7.1). Such species are responding to climate change in a predictable way having maintained within consistent climate conditions during a period of recent climate change, enabling prediction of future distribution changes under climate change using SDMs under the assumption that species will continue to maintain a static climatic niche within Great Britain. For full results across all species please see Appendix B.

Table 7.1. Species identified as climate-tracking, with an increase in occupancy between periods (positive Telfer Index) and statistically similar climatic niche between periods (niche conservatism).

Species	Common name	Type	Order	Telfer index	Niche overlap
Aeshna mixta	Migrant Hawker	Dragonfly	Odonata	1.74	0.24
Anax imperator	Emperor Dragonfly	Dragonfly	Odonata	1.68	0.35
Orthetrum cancellatum	Black-tailed Skimmer	Dragonfly	Odonata	1.30	0.26
Aeshna cyanea	Southern Hawker	Dragonfly	Odonata	0.65	0.34
Sympetrum striolatum	Common Darter	Dragonfly	Odonata	0.61	0.37
Libellula depressa	Broad-bodied Chaser	Dragonfly	Odonata	0.57	0.41
Calopteryx splendens	Banded Demoiselle	Damselfly	Odonata	0.48	0.34
Somatochlora arctica	Northern Emerald	Dragonfly	Odonata	0.44	0.21
Platycnemis pennipes	White-legged Damselfly	Damselfly	Odonata	0.23	0.29
Andrena cineraria	Ashy Mining Bee	Mining Bee	Hymenoptera	2.67	0.24
Andricus quercuscalicis	Knopper Gall Wasp	Gall Wasp	Hymenoptera	1.93	0.16
Cerceris arenaria	Sand Tailed Digger Wasp	Digger Wasp	Hymenoptera	1.64	0.15
Bombus soroeensis	Broken-belted Bumblebee	Bumblebee	Hymenoptera	1.02	0.07
Nomada panzer	Panzer's Nomad Bee	Nomad Bee	Hymenoptera	0.90	0.24
Formica fusca	Silky Ant	Ant	Hymenoptera	0.79	0.12
Abia fasciata	Banded Clubhorn	Sawfly	Hymenoptera	0.67	0.13
Tenthredo arcuate	Yellow-sided Clover Sawfly	Sawfly	Hymenoptera	0.67	0.19
Neuroterus quercusbaccarum	Spangle Gall Wasp	Gall Wasp	Hymenoptera	0.53	0.22
Neuroterus numismalis	Silk Button Gall Wasp	Gall Wasp	Hymenoptera	0.52	0.28
Andrena fulva	Tawny Mining Bee	Mining Bee	Hymenoptera	0.49	0.27
Lasioglossum rufitarse	Rufous-footed Furrow Bee	Furrow Bee	Hymenoptera	0.43	0.25
Vespula vulgaris	European Wasp	Yellowjacket	Hymenoptera	0.32	0.20
Andrena Bucephala	Big-Headed Mining Bee	Mining Bee	Hymenoptera	0.20	0.20
Crossocerus dimidiatus	Blunt Tailed Digger Wasp	Digger Wasp	Hymenoptera	0.05	0.11

7.4.2. Projection of species' future distribution

Model performance varied greatly between species, but generally with the Odonata SDMs outperforming Hymenoptera. Table 7.2 summarizes each evaluation metric, averaged across all model validation runs which were included in the final ensemble model projections (TSS > 0.4); their contribution towards habitat suitability was weighted according to model performance. Therefore, these models should be sufficiently able to discriminate suitable areas from unsuitable (Soultan & Safi, 2017). Four Odonata models had average TSS score above 0.8 deemed as excellent (Zhang et al., 2015), the remaining two Odonata species as well four Hymenoptera species had TSS scores above 0.6 (useful to excellent, Komac et al., 2016), with the remaining Hymenoptera above 0.4 (useful, Zhang et al., 2015; Soultan & Safi, 2017).

Table 7.2. Species distribution model performance averaged across all model algorithms and each model run. Note that in final ensemble models only each individual model with TSS > 0.4 contributed to predictions.

Species	Order	TSS	AUC	Accuracy	Bias	Kappa
Aeshna cyanea	Odonata	0.80	0.96	0.91	0.97	0.70
Anax imperator	Odonata	0.85	0.97	0.94	0.99	0.81
Calopteryx splendens	Odonata	0.80	0.96	0.92	0.99	0.78
Libellula depressa	Odonata	0.83	0.97	0.94	0.99	0.81
Platycnemis pennipes	Odonata	0.73	0.92	0.86	0.95	0.72
Sympetrum striolatum	Odonata	0.77	0.94	0.91	0.98	0.67
Andrena cineraria	Hymenoptera	0.70	0.91	0.86	0.98	0.69
Andrena fulva	Hymenoptera	0.72	0.93	0.88	0.98	0.73
Andricus quercuscalicis	Hymenoptera	0.45	0.79	0.74	0.90	0.41
Bombus soroeensis	Hymenoptera	0.57	0.86	0.85	0.85	0.53
Cerceris arenaria	Hymenoptera	0.66	0.91	0.83	0.94	0.63
Crossocerus dimidiatus	Hymenoptera	0.43	0.75	0.85	0.44	0.24
Formica fusca	Hymenoptera	0.61	0.88	0.80	0.95	0.60
Neuroterus numismalis	Hymenoptera	0.45	0.79	0.76	0.90	0.43
Neuroterus quercusbaccarum	Hymenoptera	0.42	0.78	0.73	0.90	0.41
Nomada panzer	Hymenoptera	0.44	0.80	0.74	0.92	0.43
Vespula vulgaris	Hymenoptera	0.57	0.87	0.82	0.98	0.55

For Odonata the total area of new habitat predicted for 2020-2040 varied between approximately 35,200 km^2 (26.3% increase) for the Emperor Dragonfly (*Anax imperator*) to 0 for the White-legged Damselfly (*Platycnemis pennipes*) for which only an area of 25 km^2 was projected to provide suitable habitat in Great Britain by 2020-2040, none of which in localities outside of the current projected habitat area (table 7.3). For one species of Hymenoptera, the Ashy Mining Bee (*Andrena cineraria*), none of the Great Britain study area is projected to be suitable by 2040. The three Hymenoptera species with the greatest projected habitat expansion were the three gall wasps the Knopper Gall Wasp (*Andricus quercuscalicis*; 176,300 km^2, 730.4% increase), Silk Button Gall Wasp (*Neuroterus numismalis*; 156,000 km^2, 301.1% increase), and Spangle Gall Wasp (*Neuroterus quercusbaccarum*; 115,100 km^2, 219.2% increase). Overall, four (66.7%) Odonata species and seven (63.6%) Hymenoptera were projected to increase in distribution size within Great Britain under the projected future climate scenario, and even of those species which are not projected to have a net distribution increase, four (66.7%) will arrive in some new localities in Great Britain. See Appendix C for full details of each species projected distribution change.

Table 7.3. Total predicted area (km^2) of suitable habitat for each species under current and future ensemble models, expansion (newly suitable area projected for future), contraction (current area no longer suitable under future prediction) and percentage change in total suitable habitat area between current and future.

Species	Order	Current	Future	Expansion	Contraction	Change
Anax imperator	Odonata	133,587	168,779	35,192	0	26.34%
Aeshna cyanea	Odonata	142,193	171,281	29,088	0	20.46%
Sympetrum striolatum	Odonata	166,925	193,635	27,035	325	16.00%
Libellula depressa	Odonata	132,762	156,798	24,236	200	18.10%
Calopteryx splendens	Odonata	109,481	108,957	10,005	10,529	-0.48%
Platycnemis pennipes	Odonata	69,884	25	0	69,859	-99.96%
Andricus quercuscalicis	Hymenoptera	24,132	200,390	176,257	0	730.38%
Neuroterus numismalis	Hymenoptera	51,795	207,721	156,001	75	301.05%
Neuroterus quercusbaccarum	Hymenoptera	52,342	167,053	115,137	425	219.16%
Cerceris arenaria	Hymenoptera	51,592	126,606	75,014	0	145.40%
Vespula vulgaris	Hymenoptera	68,154	132,112	64,007	50	93.84%
Nomada panzer	Hymenoptera	60,277	113,628	54,251	900	88.51%
Formica fusca	Hymenoptera	81,859	133,861	52,052	50	63.53%
Andrena fulva	Hymenoptera	108,030	75,639	11,579	43,969	-29.98%
Crossocerus dimidiatus	Hymenoptera	66,857	4,673	2,724	64,908	-93.01%
Bombus soroeensis	Hymenoptera	50,635	16,755	275	34,155	-66.91%
Andrena cineraria	Hymenoptera	87,195	0	0	87,195	-100.00%

7.4.3. Employing EICAT and EICAT+

Of the 9 Odonata and 15 Hymenoptera species identified as climate-tracking, impacts were reported for 6 (66.7%) Odonata and 11 (73.3%) Hymenoptera species, and those species for which no impact was detected were labelled as data deficient (DD) following EICAT convention (table 7.4). The Web of Science literature search yielded 1,267 results across all species, of which 813 were removed during abstract screening, for 107 the full text was not freely available in English, and a further 259 results were removed during the full text screenings. This resulted in 83 remaining relevant studies which documented the impact of a species of interest on another species native to the UK under an EICAT/EICAT+ mechanism. Across these studies there were 97 individual reported impacts (Appendix D); of these only a single record was assigned a confidence level of medium, and all other observations were reported as low confidence. This was primarily due to study design (87 cases, 89.7%) where the study would not have allowed the detection of impacts at a higher or lower magnitude than the one assigned, as well as studies recorded at a different scale to the national assessment conducted (76 cases, 78.3%) or extrapolation from impacts reported in other localities (66 cases, 68.0%).

Table 7.4. EICAT and EICAT+ results with overall impact category assigned. Each impact mechanism is scored according to severity of impact as follows: massive positive impact (MV+) = 5, major positive impact (MV+) = 4, moderate positive impact (MO+) = 3, minor positive impact (MN+) = 2, minimal positive impact (ML+) = 1, data deficient (DD) = 0, minimal concern (MC) = -1, minor impact (MN) = -2, moderate impact (MO) = -3, major impact (MR) = -4, massive impact (MV) = -5.

Species	**EICAT category**	**EICAT+ category**	**Impact Mechanisms**	**EICAT score**	**EICAT+ score**	**Net score**
Abia fasciata	DD	DD		0	0	0
Andrena bucephala	DD	DD		0	0	0
Andrena cineraria	DD	ML+	Dispersal (ML+) Provision through parasitism (ML+)	0	2	2
Andrena fulva	DD	ML+	Dispersal (ML+) Provision through predation (ML+)	0	2	2
Andricus quercuscalicis	MN	MO+	Provision through parasitism (MO+) Provision of habitat (MO+) Parasitism (MN) Provision through predation (ML+) Indirect impact (ML+)	-2	8	6
Bombus soroeensis	DD	ML+	Dispersal (ML+)	0	1	1
Cerceris arenaria	MO	MR+	Provision through parasitism (MR+) Predation (MO)	-3	4	1
Crossocerus dimidiatus	MC	DD	Predation (MC)	-1	0	-1
Formica fusca	MN	MO+	Provision through mutualism (MO+) Indirect impact (MN) Dispersal (MN+) Predation (MC)	-3	8	5

			Provision through predation (ML+) Provision through parasitism (ML+) Indirect impact (ML+)			
Lasioglossum rufitarse	DD	DD		0	0	0
Neuroterus numismalis	MN	ML+	Parasitism (MN) Provision of habitat (ML)	-2	1	-1
Neuroterus quercusbaccarum	MN	ML+	Parasitism (MN) Provision through parasitism (ML) Provision of habitat (ML)	-2	2	0
Nomada panzer	DD	MR+	Dispersal (MR+)	0	4	4
Tenthredo bridgmanii	DD	DD		0	0	0
Vespula vulgaris	MO	MO+	Predation (MO) Provision through parasitism (MO+) Dispersal (MN+) Competition (MC) Disease transmission (MC)	-5	5	0
Aeshna cyanea	MO	DD	Predation (MO)	-3	0	-3
Aeshna mixta	DD	DD		0	0	0
Anax imperator	MO	DD	Predation (MO) Indirect impact (MC)	-4	0	-4
Calopteryx splendens	MN	DD	Hybridisation (MN)	-2	0	-2
Libellula depressa	MC	ML+	Predation (MC) Provision through predation (ML+)	-1	1	0
Orthetrum cancellatum	DD	DD		0	0	0

Platycnemis pennipes	DD	ML+	Provision through parasitism (ML+)	0	1	1
Somatochlora arctica	DD	DD		0	0	0
Sympetrum striolatum	MO	DD	Predation (MO)	-3	0	-3

Overall, there were 39 unique impacts across species under separate impact mechanisms with impacts being reported as the most severe category recorded under each impact mechanism (table 7.4, figure 7.1). For Hymenoptera, deleterious impacts were detected under five impact mechanisms, most commonly predation (four species), followed by parasitism (three species), with one species reported to impact other native taxa through competition, disease transmission, and indirect impacts. Positive impacts were reported under six different impact mechanisms, namely provision of trophic resources through parasitism (seven species) and dispersal facilitation through mutualism (six species). Three species also provided trophic resources through predation, three provided habitat through commensalism, two positively impacted native species through an indirect impact, and one species provided a trophic resource through mutualism. Odonata species impacted other native taxa through a narrower range of impact mechanisms than Hymenoptera, with predation by four species, and the remaining impact mechanisms only recorded for a single species (provision of trophic resources through predation and parasitism, hybridisation, and indirect impacts).

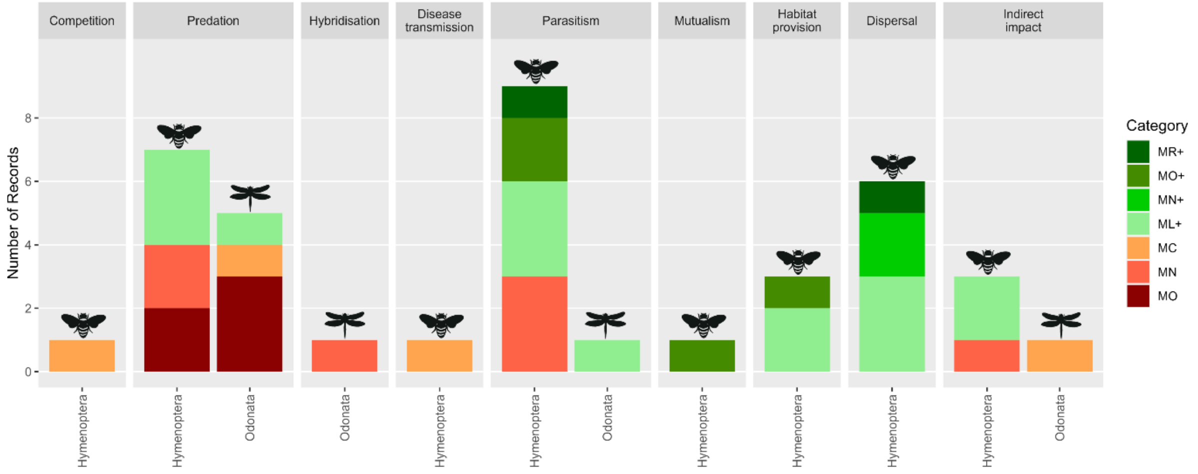


Figure 7.1. Range of impact mechanisms across Odonata and Hymenoptera species and impact severity categories: Major positive impact (MR+), Moderate positive impact (MO+), Minor positive impact (MN+), Minimal positive impact, (ML+), Minimal Concern (MC), Minor (MN), Moderate (MO).

Of the reported impacts by species, 19 (48.7%) were at the lowest category level (either Minimal Concern (MC) under EICAT or Minimal (ML+) positive impact under EICAT+), and an additional nine (23.1%) at the second lowest impact category (Minor). No impacts were reported at the highest impact level (Massive), which is defined as an impact resulting in naturally irreversible community level changes through local or global extinction of a native species under EICAT or causing a long-lasting increase in occupancy of a native species through local re-establishment or extinction prevention through EICAT+, nor were any negative impacts detected under the second highest impact level (Major) whereby impacts are reported to cause a decrease in the area of occupancy of another native taxa. The highest reported positive impact level was Major (MR+), causing a transient increase in occupancy in another native taxa, observed for two Hymenoptera species (5.1% reported impacts) – the provision of trophic resources through parasitism of the Sand Tailed Digger Wasp (*Cerceris arenaria*; Paukkunen et al., 2018) and dispersal facilitation through mutualism of the Panzer's Nomad Bee (*Nomada panzer*; Kolanowska & Jakubska-Busse, 2020). The remaining nine (23.1%) impacts reported were Moderate (MO or MO+), causing a decrease or increase in the population size of another native taxa for EICAT and EICAT+, respectively.

The lowest total negative impact score assigned to a species was minus five for the European Wasp (*Vespula vulgaris*; table 7.4), which caused a Moderate (MO) negative impact through predation of Small White caterpillars (*Pieris rapae*) and Turnip Sawfly larvae (*Athalia rosae*; Müller & Brakefield, 2003) and minimal impact through competition with the German Wasp (*Vespula germanica*; Pereira et al., 2016) and Western Honey Bee (*Apis mellifera*; Markwell et al., 1993) as well as disease transmission to the Western Honey Bee (*A. mellifera;* Remnant et al., 2021). The highest positive impact score was eight for both the Knopper Gall Wasp (*A. quercuscalicis*) and Silky Ant (*Formica fusca*). The former having a Moderate (MO+) impact through the provision of trophic resources through parasitism and the provision of habitat through commensalism for numerous species of parasitic and inquiline wasps, respectively (see table D.1), as well as Minimal (ML+) positive impact though predation by the Eurasian Blue Tit (*Cyanistes caeruleus*; Hails & Crawley, 1991) and indirectly through parasitism of the Gall Wasp's inquilines (Schonrogge et al., 1995). The latter provides a Moderate (MO+) positive impact by provision of trophic resources through mutualism with leafhopper *Dalbulus quinquenotatus* (Larsen et al., 2001), and a Minimal (ML+) positive impact by direct provision through predation by the Western Capercaillie (*Tetrao urogallus*; Borchtchevski, 1994) and Eurasian wryneck (*Jynx torquilla*; Andersen et al., 2018) and social parasitism by other ant species (*Formica polyctena, Formica sanguinea* and *Polygerus rufescens*; Mori & le Moli, 1998; Czechowski, 2006; Czechowski & Marko, 2006; Czechowski, 2007; Wlodarczyk & Szczepaniak, 2017), as well as indirectly impacting *Myrmica sabuleti* ant colonies through provision of carcasses following raids by slave-making *P. rufescens* ants (Pech, 2012).

The net impact score across species ranged from minus four for the Emperor Dragonfly (*A. imperator)* to six for the Knopper Gall Wasp (*A. quercuscalicis*). Of the 15 climate-tracking Hymenopteran species, six species (40.0%; *Abia fasciata, Andrena bucephala, Lasioglossum rufitarse, N. quercusbaccarum, Tenthredo bridgmanii* and *V. vulgaris*) were assigned a net impact score of zero (no detected impact or equivalent positive and negative impact scores), seven (46.7%; *A. cineraria, Andrena fulva; A. quercuscalicis; Bombus soroeensis, C. arenaria, F. fusca* and *N. panzer*) had a net positive impact and two a net negative impact (13.3%; *C. dimidiatus* and *N. numismalis*). For the nine Odonata, four (44.4%; *Aeshna mixta, Libellula depressa, Orthetrum cancellatum* and *Somatochlora arctica*) had a no net impact, four (44.4%; *Aeshna cyanea, A. imperator, Calopteryx splendens* and *Sympetrum striolatum*) a net negative impact, and only a single species (11.1%; *P. pennipes*) a net positive impact.

In total 99 other native species were reported to be impacted by the climate-tracking Hymenoptera, and 18 by the Odonata (figure 7.2). These impacted species were largely other Insecta, 70.7% (N = 70) for Hymenoptera and 50.0% (N = 9) for Odonata, and dominated by other species within the same taxonomic group, 38.4% (N = 38) for Hymenoptera and 33.3% (N = 6) for Odonata; though these intragroup interactions where largely positive for Hymenoptera in many cases provision of trophic resources and habitat by the gall wasps, and negative for Odonata, namely by predation. Hymenoptera climate-trackers also impacted several Coleoptera species, as a large number of weevil species were identified as prey for Hymenoptera. Several plant species also gained from dispersal facilitation by Hymenoptera climate-trackers, and the other non-insect groups impacted were four bird species identified as predators of Hymenoptera species. Odonata impacted other aquatic taxa primarily though predation (either as predators or prey) including four fish species, three frog species, and two Crustacea – a water flea and Ostracod.

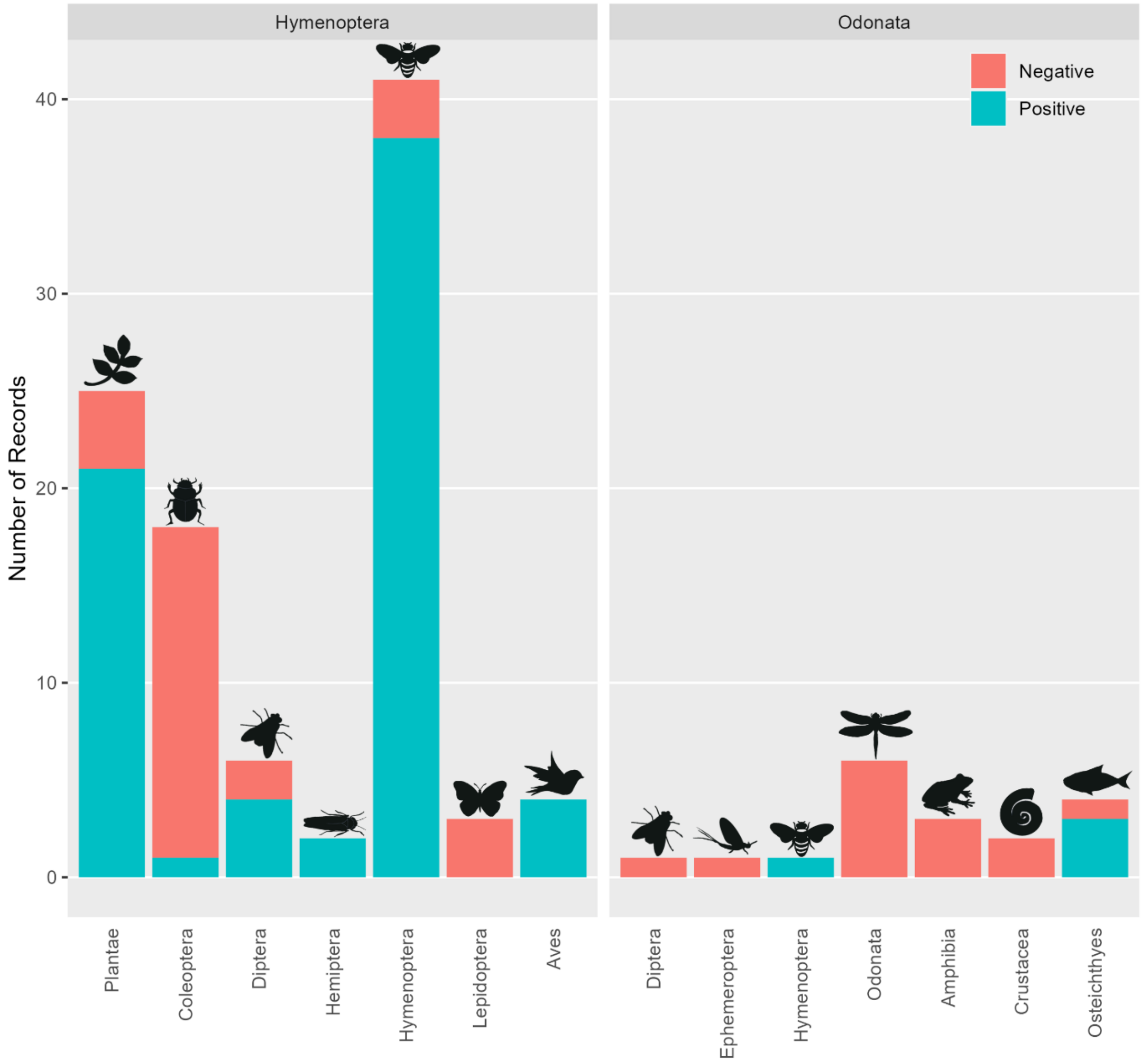


Figure 7.2. Number of native species impacted by the Odonata and Hymenoptera study species assessed under EICAT and EICAT+.

7.4.4. Spatial impact assessment

A spatial assessment of the potential impact of climate-tracking species was conducted, including those species for which an impact was identified under EICAT/EICAT+ that were projected to arrive in new localities within Great Britain under future climate change (i.e. expansion > 0 km^2), consisting of five Odonata (*A. cyanea, A. imperator, C. splendens, L. depressa* and *S. striolatum*; figure 7.3) and ten Hymenoptera species (*A. fulva, A. quercuscalicis, B. soroeensis, C. arenaria, C. dimidiatus, F. fusca, N. numismalis, N. quercusbaccarum, N. panzer* and *V. vulgaris*; figure 7.4). For Odonata, all species are currently distributed in the Southeast and primarily projected to expand into new areas in the North, Southwest, and Wales, whilst largely retaining their current occupancy in the Southeast, as such there are no new arrivals predicted for much of England. Projected newly suitable habitat is largely in Scotland, particularly on the coast, the south, east and the islands, however, largely excluding the highlands. One species, the Common Darter (*S. striolatum*), is projected to be found sporadically in the north-most Scotland and at higher altitudes than the other Odonates; this species could potentially impact the ostracod *Eucypris virens* through predation (Schmit et al., 2012). In England the areas with the greatest numbers of new arrivals are at the highest altitudes such as in Wales, Dartmoor, Exmoor, the North York Moors, and the Peak District, whilst species are excluded from the highest altitudes further north such as the Yorkshire Dales and the North Pennines. In South Wales and the Southwest there are areas where only one species, the Banded Demoiselle (*Calopteryx splendens*), is projected to expand, which could negatively impact the Beautiful Demoiselle (*Calopteryx virgo*) through hybridisation (Tynkkynen et al., 2008; Tynkkynen et al., 2009; Wellenreuther et al., 2010). Total negative and net impacts largely follow the same pattern, being most concentrated where the greatest number of new species arrivals are projected as there was only a small variation in the impact scores of species. In terms of positive impacts, only a single species with a positive impact was projected to gain suitable area under climate change, the Broad-bodied Chaser (*Libellula depressa*) with a score of 1 for minimal provision of trophic resources through predation by perch (*perca fluviatils*), rudd (*scardinius erythrophthalmus*), and gudgeon (*gobio gobio*; Wohlfahrt et al., 2006).

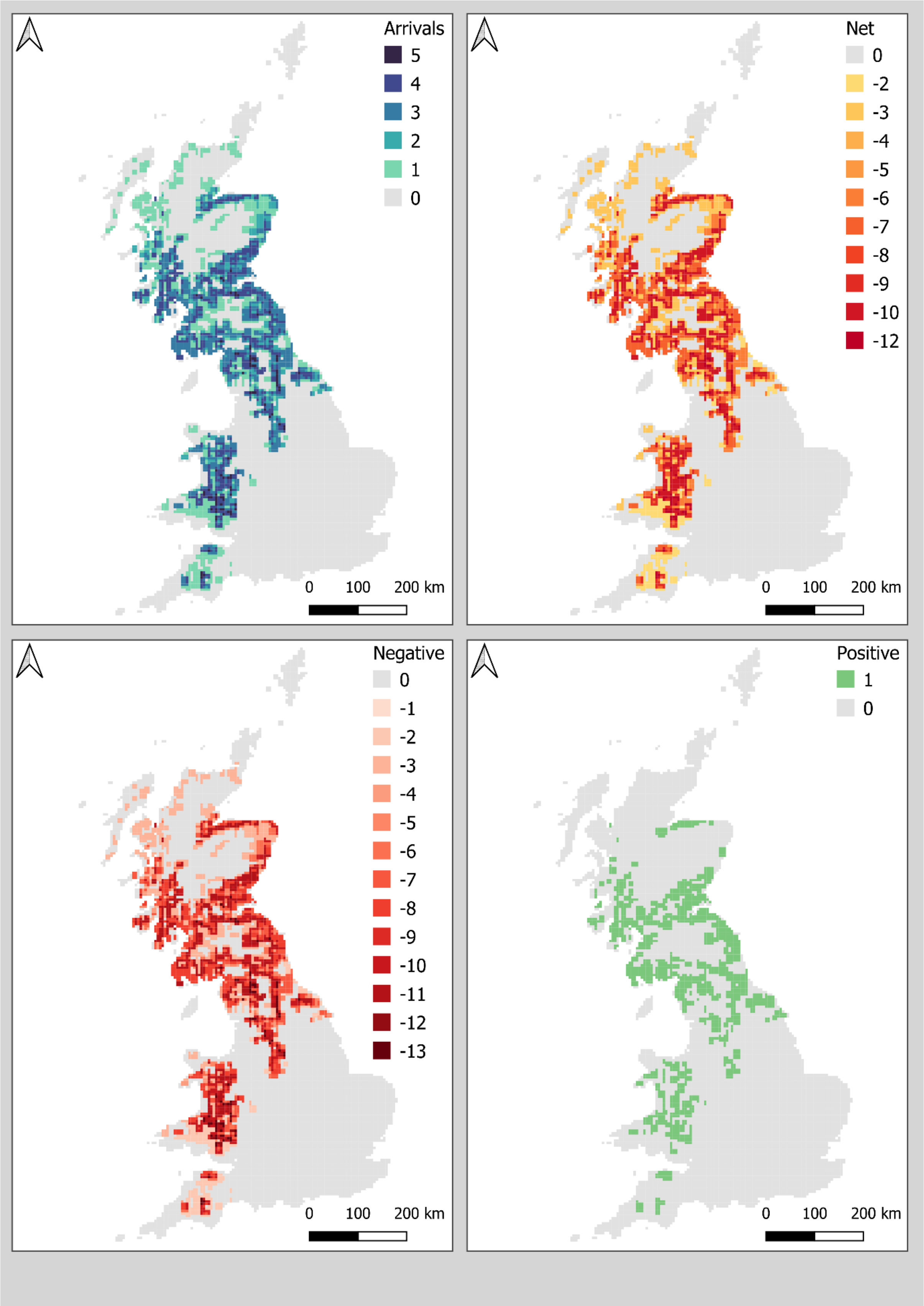


Figure 7.3. Spatial impact assessment of Odonata species of interest, showing total number of species arriving in each 5km grid as well as cumulative positive, negative, and net impacts.

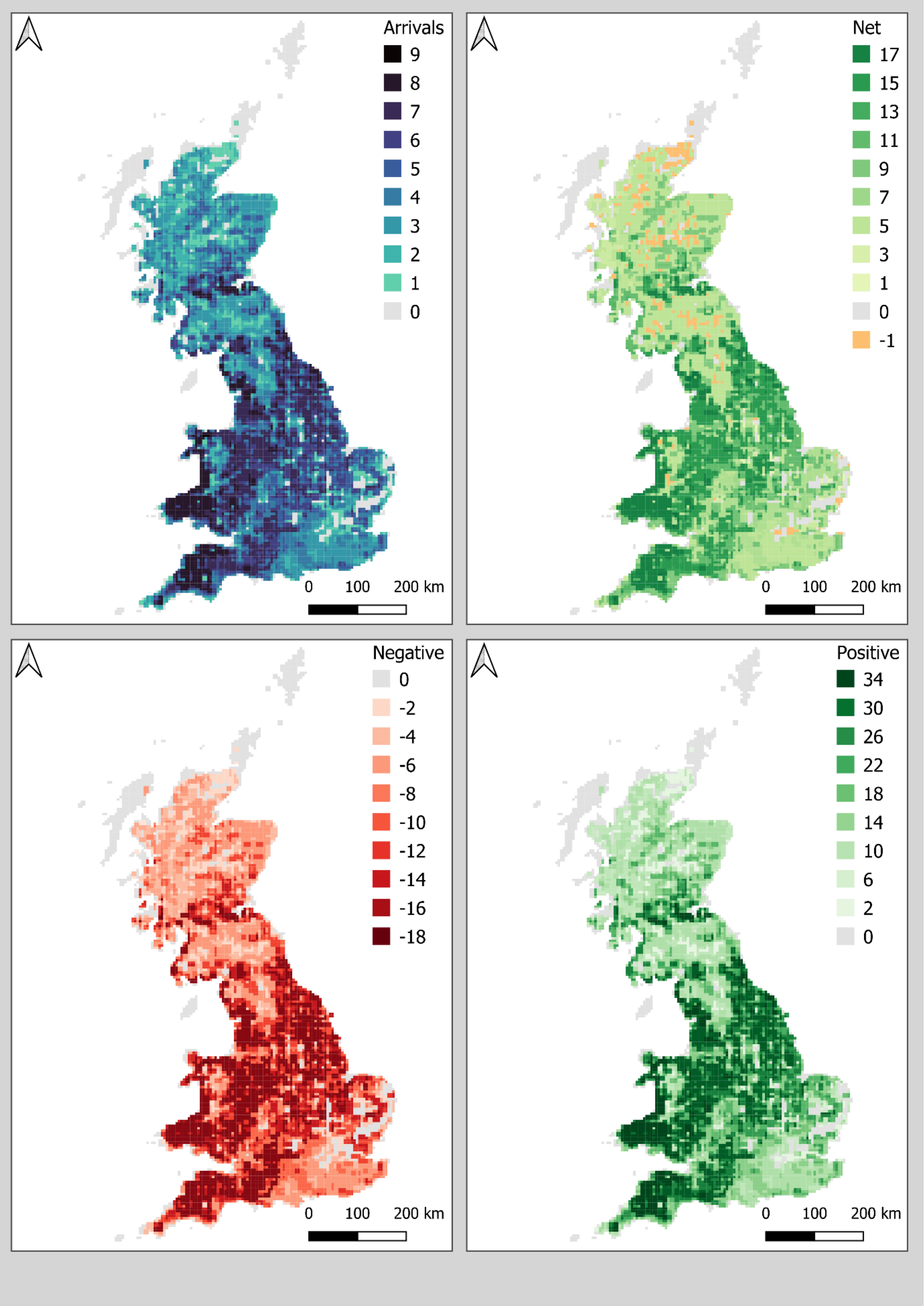


Figure 7.4. Spatial impact assessment of Hymenoptera species of interest, showing total number of species arriving in each 5km grid as well as cumulative positive, negative, and net impacts.

For Hymenoptera, projected potential species arrivals are more widespread throughout Great Britain, but with the greatest increased suitability for species in South Wales, parts of the Southwest and much of England and the lowest species arrivals at the highest altitudes. As with the Odonata, the East and Southeast have the smallest number of new arrivals, as species are largely already present. At least one species of the ten Hymenopteran study species modelled, is projected to arrive throughout almost the entire Great Britain study area under climate change projections, other than the Outer Hebrides, Orkney Islands, Shetland Islands, the Isle of Man, as well as some areas in the East and Southeast. Although there are fewer projected species arrivals in Scotland and at high altitudes than other areas, these areas will still be impacted. The three gall wasps (*A. quercuscalicis*, *N. numismalis, N. quercusbaccarum*) are projected to undergo the greatest expansion under projected climate change, with projected suitable habitat across Great Britain; these species negatively impact our native oaks through gall formation, whilst providing trophic resources to other parasitic and inquiline Hymenoptera species (see table D.1 for full details). The Sand Tailed Digger Wasp (*C. arenaria*) is also projected expand substantially under climate change (with an additional 54,251 km^2 potential suitable habitat across England and Southern Scotland); this species has potential for Moderate (MO) deleterious impacts through predation of several weevil species (Polidori et al., 2005) and Major (MR+) positive impacts through parasitism by the Cuckoo Wasp (*Hedychrum nobile*; Paukkunen et al., 2018). The species with the smallest expansion of predicted suitable habitat are the Broken-belted Bumblebee (*Bombus soroeensis*) and the Blunt Tailed Digger Wasp (*Crossocerus dimidiatus*), both projected to expand into the Scottish Highlands and West Scotland, the former classified as having a Minimal (ML+) positive impact through dispersal facilitation of the Bog Bilberry (*Vaccinium uliginosum*; Mayer et al., 2012) and the latter classified as Minimal Concern (MC) through predation of two root-maggot fly species (*Hylemya variata* and *H. strenua*; Otoole & Raw, 1978). Total cumulative positive impact scores were higher than total negative scores, and so the net impact is largely positive throughout Great Britain, however, there are some isolated points where negative impacts exceed positive, such as in Scotland and high altitudes which is driven by the arrival of the Silk Button Gall Wasp (*N. numismalis*) and the Blunt Tailed Digger Wasp (*C. dimidiatus*) which have net negative impact scores in localities that are not predicted to be suitable for those species with positive impact scores.

7.5. Discussion

Here it is demonstrated for the first time that the EICAT and EICAT+ frameworks for invasive species provide a potential methodology to review the possible impact of climate-tracking species, albeit with some reservations. By conducting a thorough literature review on the Web of Science, 83 relevant studies were collated across the study species, constituting a total of 97 individual reported impacts and 39 unique impacts across species under separate impact mechanisms with a total of 117 impacted native species. The Odonata and Hymenoptera had distinct spatial patterns of projected impacts under a future climate scenario, with the total net impact score according to the severity of impact ascribed across each mechanism being primarily negative for Odonata and positive for Hymenoptera.

The notion that structures within invasion biology could potentially be adjusted to focus on the assessment of species that are shifting due to climate change was introduced by Wallingford et al (2020). Urban (2020) expressed concern that the application of invasive perspectives for climate-tracking species may threaten conservation efforts through an association with the language and culture of eradication within invasion biology. However, by employing both the EICAT framework to assess negative impacts as well as EICAT+ for positive impacts, impacts could be measured objectively as positive or negative depending on whether they induce an increase or decrease in a particular measurable attribute, rather than according to a subjective perception of harmful impacts as those deemed as unfavourable (Vimercati et al., 2022).

Employing an invasive framework for climate tracking species might however be further limited due to other fundamental differences (see Urban, 2020 for a summary of these distinctions). As an example for this case study, predation was the most common mechanism through which species negatively affected other native species, such as the Southern Hawker (*Aeshna cyanea*) and Emperor Dragonfly (*A. imperator*) both predate on other Odonates as well as tadpoles. For invasive species, it is theorised under the 'naïve prey' hypothesis that lacking an evolutionary history with alien predators, native prey will be particularly susceptible to predation owing to ineffective antipredator responses to novel predators. Therefore, climate shifting species may have reduced deleterious ecological impacts having a previous evolutionary history with species in the recipient environment or gene flow from nearby populations (Urban, 2020). For example, Gazzola et al. (2023) demonstrated an anti-predator response by Agile Frog tadpoles (*Rana dalmatina*) to native Southern Hawker larvae (*A. cyanea*), even when tadpoles had not been conditioned to predator cues as embryos (controls); these tadpoles were capable of associating environmental signals with predation threat for the predators to which they shared a long evolutionary history, however, the same was not the case for alien Red Swamp Crayfish (*Procambarus clarkii*). Similarly, Agile Frog (*R. dalmatina*) populations were found to exhibit a lack of defensive responses against Red Swamp Crayfish (*P. clarkii*) by Gazzola et al. (2018), whereas tadpoles significantly reduced activity and visibility in the presence of tadpole-fed Emperor Dragonfly larvae (*A. imperator*). On the other hand, there is large variation in the establishment success of invasive species with some examples where alien species instead incur a 'novelty disadvantage' being naïve about the novel prey and predators in the recipient community resulting in poor foraging success or limited antipredator response to top predators (Sih et al., 2010); this is theorised to be most likely when the alien species source and invaded community is fundamentally dissimilar. Species expanding into new areas with climate change, having a fundamentally similar community and shared evolutionary history might prove an advantage, such as was found to be the case by Zebsa et al (2022) where familiar low-elevation predators (*A. cyanea*) and unfamiliar high-elevation predators (*Aeshna juncea*) induced similar antipredator responses by range shifting Common Darter Dragonflies (*S. striolatum*). This study emphasises the need for direct analysis of species redistribution under climate change to determine the extent and severity of the ecological impacts to other native species and the influence of novel biotic interactions. Future advancements to our methodology could incorporate information on the geographic co-occurrence and niche overlap between climate-trackers and affected native species, under current and future climate scenarios, such as to provide a measure of the degree of 'novelty' of resulting biotic interactions.

Although adopted by the IUCN, impact categorisation methods have been criticised for a lack of consistency as outcomes have been found to differ between assessors (Strubbe et al., 2011; Kumschick et al., 2017) as well as higher impact scores resulting from more intensive and costly literature searches (Measey et al., 2020). Furthermore, there is often insufficient information available to categorise impacts and assessments are frequently made with a low confidence rating (Evans et al., 2016). Employing these methods for climate-tracking species inflates these uncertainties as well as introduces further limitations. For EICAT/EICAT+ there is an emphasis that the impacts of invasive species must be measured within the specific context under which impacts are assessed, for example a national scale assessment would include only evidence from that specific country, thereby assessments are grounded on specific and relevant scientific studies often reporting impacts which have already occurred and been quantified (Volery et al., 2020). This approach is possible for invasive species that have already been defined by the impact they are causing to native taxa within their non-native range, and therefore a body of literature is available. To assess climate-tracking species on a national scale, a less focused assessment was undertaken where the only stipulation for the inclusion of relevant literature was information regarding an impact to a native UK species through an EICAT/EICAT+ mechanism. Therefore, almost every impact was reported with low confidence due to the extrapolation of information from another locality and because most studies included would not have allowed for the detection of impacts at a higher or lower magnitude having not been specifically aimed at measuring impact severity. Furthermore, impacts were also typically measured on a low spatial scale compared to the national scale of the assessment, adding a further source of uncertainty.

This procedure assigns a confidence level for each individual impact report to capture different sources of uncertainty, however, there are other types of biases which can occur beyond the individual report level which are not captured by the confidence score (Probert et al., 2020). For example, there can be bias present in the existing data or other biases introduced during the data collection and assessment process. This approach is likely biased towards detecting greater adverse impacts of species at higher trophic levels, such as assigning high severity impact scores to predators without recognising the potential benefits of predation for controlling prey populations. Under EICAT any impact that causes a decline in population size is categorised as Moderate (MO), whilst EICAT+ only recognises a positive impact through predation whereby the functional roles are reversed such as the non-native species provides prey for another native species. Publication bias has also previously been shown to affect invasive impact assessments, with non-significant or null results being less likely to be submitted for publication (Dueñas et al., 2018). For this assessment where potential impacts have been assessed but no change to the performance of natives is detected a minimal impact category is assigned; though relatively numerous within this dataset, these might only represent a subset of work for which no measurable change was observed as such studies might be less likely to be published. Moreover, as with other GISS schemes a precautionary principle is adopted whereby each species is classified at the highest magnitude level reached, so even where literature may indicate negligible impacts for several native species, a higher impact score might be assigned if a single study demonstrates a single impact at a greater magnitude. Other alternative metrics might be more appropriate and should be explored, particularly for implementation with climate-tracking species, for example calculating a weighted score according to the amount of support within literature for impacts at different magnitudes. The confidence level could likewise be ascribed according to coherence within the evidence gathered during the literature survey. Such methods, however, would require sufficient relevant studies documenting these interactions which are not currently uniformly available across climate-tracking species.

The methodology employed to assign impact scores to each species also warrants further examination. Here a score is given corresponding to the impact category under each impact mechanism for each species which are summed to investigate the potential geographical distribution of positive and adverse impacts in Britain. This method aims to give an indication of the impact severity according to the number of species arrivals and magnitude of impact under a specific number of mechanisms, however, does not capture the nuances of different potential scenarios. For example, one species which influences several native species through a range of different mechanisms but with a low magnitude might potentially be more damaging than a high magnitude impact through a single interaction, yet both cases could receive the same score. On the other hand, even when species cause only a negligible level of impact or effect to the performance of a native species they are categorised at least as Minimal Concern (MC; IUCN, 2020a). This causes a further limitation of a semi-quantitative approach as three species arriving at a specific locality that cause minimal impact under EICAT would be given a cumulative score of negative three and a single species arriving with a Moderate (MO) impact, causing a decline in the population size of a native species, would be assigned the same score. Alternative methods should be explored such as to incorporate a measure of how many native taxa are impacted as well as a variety of interactions. Nevertheless, the technique applied here provides an example of how a GISS assessment can be instantaneously translated into a spatial impact assessment through a simple scoring system.

There are further limitations to this approach with regards to the uncertainty in the number of new species projected to arrive in an area through the implementation of SDMs, as areas predicted to be climatically suitable in the future might not necessarily be occupied by species. Here the RCP8.5 projection was employed to show a worst-case scenario, defined by radiative forcing levels of 8.5 watts per square meter (Van Vuuren et al., 2011); therefore, these results are based on a single plausible scenario of a potential climate future for illustrative purposes. Additionally, these projections are based on climate average indices and bioclimatic variables without consideration of climate extremes which are important in shaping species distributions (Huang et al., 2017), however, these were not incorporated as climate extreme indices cannot be predicted at a comparable resolution to the average climate projections available. Furthermore, these SDMs give only an indication of potential habitat suitability according to climate changes, without consideration of other environmental variables that shape species' distributions and so projected range shifts will be moderated by habitat availability as well as other non-climate factors such as land use change. Information on the coverage of broadleaf woodland was the only other environmental predictor employed in the SDMs, deemed particularly important for the gall species, however, these species are explicitly linked to oak and so species level information would be more appropriate though not available. Moreover, the potential changes in broadleaf cover under the future climate scenario was not incorporated into projections. As such, the large expansion of habitat suitability predicted for the gall wasps is likely to be an overestimation as will be mediated by changes in broadleaf woodland, availability of oak specifically, as well as other habitat factors.

Species' projected distribution changes will also vary according to their dispersal ability which is not incorporated into SDM projections thereby adding to the uncertainty regarding the number of predicted species arrivals for a given area. Moreover, dispersal capacity is also an important factor for the impact of invasive species (Wallingford et al., 2020) and might likewise influence the adverse impact of climate-tracking species. In this case, it is unlikely that the inclusion of dispersal would greatly influence findings as the focal taxa were chosen according to their dispersal abilities (Cancellario et al., 2022), however, it has been shown that incorporating dispersal measures or scenarios into SDM provides more robust projections (Bateman et al., 2013). Specific information about the dispersal capacity of the study species was not directly available, however, a potential dispersal distance can be calculated for Odonata taking into account the duration of larval development and the given time period over which projections are made (Jaeschke et al., 2013). Without information on observed dispersal distances, these can be estimated based on expert opinion (Harabiš & Dolný, 2011), indicating that the Odonata study species could disperse between 300 km and 1400 km depending on dispersal capacity, a distance sufficient to correspond with projected expansions. For Hymenoptera, dispersal abilities are largely unknown or uncertain (Sirois-Delisle & Kerr, 2018), therefore dispersal limitations were not incorporated into projections and the methodology applied was consistent for both taxonomic groups. Evidence does indicate that bumblebees can shift between 3 and 5 km a year, sometimes as much as 10 km (Sirois-Delisle & Kerr, 2018), whilst Vespula species can spread up to 40 km per year (Lester & Beggs, 2019) and the Silky Ant (*F. Fusca*) has the potential to travel 57 km in up to 250 minutes depending on wind conditions (Johansson et al., 2018). Though the projected expansions of focal climate-tracking species under the given timescale are broadly feasible; in applying this methodology for other species it is valuable to consider dispersal capability during model projections. Moreover, it remains that other factors such as habitat availability and topographic features might indeed prohibit species successful expansion. It should also be noted that the purpose of the spatial assessment will greatly influence the chosen methodology employed. Here we reveal areas that are predicted to be climatically suitable for focal species under predicted climate change scenarios, including primarily climatic variables. This method might, for example, be useful for detecting potential future climate suitable areas for the managed relocation of species, necessitating an impact assessment for which dispersal capability would be irrelevant.

Further uncertainty in the arrival of species projected under climate change results from poor model projection for some species. Although models with a TSS score above 0.4 which were implemented here have been classified as useful (Zhang et al., 2015; Soultan & Safi, 2017), such scores are by no means optimal and introduce a significant amount of uncertainty in model results. Introducing further parameters to better capture habitat suitability, or only projecting impacts for a subset species with high performing models would provide a more accurate assessment. Hymenoptera projections are more uncertain, with the gall wasps and the European Wasp (*V. vulgaris*) performing particularly poorly, likely influencing results as these species were predicted to have a large degree of expansion under climate change with a high impact. Accordingly, the projected spatial impact assessment for Hymenoptera may be exaggerated and requires further investigation.

A further limitation of the spatial impact assessment is that that the distribution of impacted native species is not considered. This causes several uncertainties, as species that are shifting into new localities might not necessarily encounter the native taxa which they are reported to impact, meaning any potential benefit or impact might not be realised even if the study species shift. On the other hand, these climate-tracking species might already co-occur with these other native species, with their interactions simply forming a natural part of the community composition and normal ecosystem functioning; whereas the deleterious impacts of climate-tracking species are likely to result from novel biotic interactions (Pecl et al., 2017), such as when species shift at different rates from their enemies or across large distances (Wallingford et al., 2020). Here there is potential for overestimating impact severity of climate-tracking species, as they might encounter the same species as they expand into new areas. This methodology disregards the complexity through which species impact recipient environments under climate change. For example, a new predator arriving at a given locality will not necessarily cause an adverse impact, particularly when replacing another predator that has shifted elsewhere. Deleterious impacts could also occur for climate-tracking species that have a beneficial interaction with other native species which are contracting or due to changes of abundance within a species current geographic range, both of which are distributional changes which are omitted from this study. Additionally, this study focusses on the impacts of species which have exhibited niche conservatism during past climate change, forming an important assumption underpinning the accurate application of SDM (**chapter 5**; Broennimann et al., 2011), however, thereby excluding the impacts of other range-expanding species. Although model projections might be tentative, predicting potential impacts for niche flexible species would be valuable to assess whether impacts are likely to be greater or less than the chosen subset of species. An alternative approach could be to include all species within a taxonomic group, specifying the level of uncertainty according to model performance and degree of climate niche consistency during past warming. Given that there is currently no framework in place to assess the impacts of climate-tracking species, this study demonstrates a potential approach for a subset of species which gathers available information on interactions that could potentially lead to impacts and maps these according to predicted expansion of climate-tracking species. The methodologies employed here could be developed further to incorporate greater complexity or capture different scenarios, such as to include the changing distribution of impacted species or to investigate the impacts of species contractions or niche flexible species.

In terms of the taxonomic groups chosen to illustrate this methodology, it was hypothesised that Hymenoptera would be a more problematic group, with greater negative consequences than Odonata. Hymenoptera were found to impact a greater number of native species, through a broader range of mechanisms, with a greater total value of cumulative negative scores than for Odonata. On the other hand, cumulative negative scores were greater for Odonata per species, with a similar number of native species impacted per Odonata species investigated. Moreover, Odonata had a greater proportion of species with negative impacts, and a greater number of species with the highest severity of impact reported (Moderate). Therefore, our results suggest that taxonomic groups with a greater prevalence of invasive species are not necessarily more likely to cause adverse impacts when shifting due to climate change. In fact, there are no Odonata species listed on the GISD (http://www.issg.org/database), yet the results of the EICAT/EICAT+ assessments employed here suggest that of the species investigated many could adversely impact other native species under climate change, with three species (*A. cyanea, A. imperator*, and *S. striolatum*) categorised as Moderate (MO), causing population declines in other native species through predation.

For Hymenoptera, as with Odonata, the greatest severity of impact occurred through predation; this followed expectations as it was theorised that impact mechanisms by climate-tracking species might parallel those through which invasive Hymenoptera species impact native species. There are 28 invasive Hymenoptera species reported on the GISD (http://www.issg.org/database), for which impacts have been detected through EICAT under the mechanisms: competition, predation, transmission of disease, parasitism, and poisoning/toxicity. Indeed, the reported potential impacts by the study species investigated were confined to those mechanisms, with only poisoning/toxicity not represented, therefore providing evidence that the impact mechanisms of invasive species within a taxonomic group does provide some indication of the likely adverse interactions between climate-tracking species and other natives. However, the Hymenoptera species on the GISD are dominated by Formicidae (ants; 67.9%) for which these impact mechanisms are represented, and other than the Sirex Woodwasp (*Sirex noctilio*: Siricidae) for which parasitism is reported, EICAT impact mechanisms have not been reported for the other Hymenopteran families within the GISD. Here, it was the Sand Tailed Digger Wasp (*C. arenaria*), the Blunt Tailed Digger Wasp (*C. dimidiatus*), the European Wasp (*V. vulgaris*) and the Silky Ant (*F. fusca*) that impacted other natives through predation, and the three gall wasp species (*A. quercuscalicis, N. numismalis* and *N. quercusbaccarum*) impacted native oaks through parasitism. Impacts through the other mechanisms – competition and disease transmission, was only reported for the European Wasp (*V. vulgaris*). Vespidae are also well represented within Hymenoptera on the GISD (5 of the total 28 Hymenoptera species), with the European Wasp (*V. vulgaris*) itself being invasive outside the UK. In fact, some of the EICAT reports collated here are for observations of the European Wasp (*V. vulgaris*) within its non-native range, such as competition with the German Wasp (*V. germanica*) in Argentina where both species are invasive (Pereira et al., 2016), and competition with the Western Honey Bee (*A. mellifera*) for honeydew food resources in New Zealand (Markwell et al., 1993); such results can only be extrapolated to the UK with low confidence.

In terms of the spatial impact assessment, potential impacts for Hymenoptera are more geographically spread than for Odonata, however, both groups follow broadly similar patterns with the spread of our current natives under future climate change towards higher latitudes and altitudes, consistent with climate warming and previously reported shifts by species (Lenoir & Svenning, 2015). For Odonata, all the species investigated currently have a southeasterly distribution and so when mapping the potential impact of species expansion under climate change the Southeast appears to be unaffected. It should therefore be noted that with continued climate change it is likely that the Southeast will be the first area to be colonised by new Odonates from the Continent, such has already been observed for the Small Red-eyed Damselfly (*Erythromma viridulum*) which has been reported since 1999 (Cranston et al., 2023). As with invasive species, such new arrivals are more likely to form novel biotic interactions and might therefore be theorised to have a greater harmful impact to recipient environments.

Despite the limitations to this approach, as there few studies directly reporting the impact of climate-tracking species' on recipient environments, employing EICAT/EICAT+ offers a novel, useful, and targeted approach for gathering potential information sources, reviewing the types of potential impacts for recipient environments by focusing on the EICAT/EICAT+ mechanisms, as well as a process to assign a semi-quantitative impact score that indicates the potential impact intensity. It is emphasised within the guidelines of EICAT itself that it is not intended as a risk assessment, nor can its output alone inform management actions (IUCN, 2020a), rather employing EICAT/EICAT+ provides a consistent and comparable manner to assess impacts that can inform risk assessments and prioritisation of management activities alongside other information sources. It is difficult to measure the success in employing EICAT/EICAT+ for climate-shifting species without any concrete evidence of reported impacts in the specific localities to which species are arriving under climate change. It is interesting, though unsurprising that the species which was found to cause the greatest potential negative impact was the European Wasp (*V. vulgaris*), a species that is invasive outside the UK, though difficult to attribute this finding to an actual greater propensity for deleterious impacts or resulting from a greater degree of relevant scientific literature.

8. Discussion

This discussion draws together the separate chapters of this thesis to provide an overview of how this work furthers the field of species redistribution. Firstly, the key contributions of this research are highlighted emphasising the specific advancements made by each data chapter; this is followed by a discussion of the principal limitations of the overall approach taken, and then recommendations are made for additional research which would address remaining knowledge gaps, and which are guided by the findings discovered during this research. Finally, the primary discoveries made through this research are considered within the broader context of the biodiversity and climate crises and the conservation challenges posed by species on the move with climate change.

8.1. Contribution to the field

This thesis contributes several important advances to redistribution science, a field of study which encompasses both the distribution changes by species due to climate change as well as the ecological and societal consequences of such shifts (Bonebrake et al., 2017). Expanding current understanding within this field is imperative for conservation efforts to reduce biodiversity loss and enhance resilience to climate change; particularly as species movements due to climate changes pose significant challenges to conventional conservation strategies (Cranston et al., 2023).

One key finding of this work, is the importance of employing diverse occurrence data types when mapping a species distribution and evaluating their environmental niche. **Chapter 5** illustrates that the projections of habitat suitability of a specific species, the Banded Demoiselle (*Calopteryx splendens*), differ between different types of biological records – traditional, citizen science, and social media. In particular, including social media records which where manually extracted from existing Facebook and Twitter content as well as from Flickr using the Flickr Application Programming Interface (API), captured the Banded Demoiselle (*C. splendens*) at localities within Great Britian which were not represented by other more numerous and widely employed data types. When these social media observations are excluded, both the species' environmental niche and predicted habitat suitability under Species Distribution Model (SDM) projections are underestimated. These findings demonstrate that the integration of diverse data types captures new information regarding species' environmental niche and current distribution. Moreover, with the potential for rapid flagging of species arrivals in new localities (**chapter 5**; Pettorelli et al., 2019), social media content is likely to be an increasingly important source of biological records, particularly in instances where alternative data is lacking (Di Minin et al., 2015; Allain, 2019).

Correlative SDMs are widely employed to predict species future distributions under climate change; this has become a prevalent aspect of explanatory ecology within redistribution science. Such approaches assume that species typically track climate changes geographically by shifting to remain within their optimal climate conditions – i.e. the theory of niche conservatism (Ralston et al., 2017) – the tendency of a species' niche to remain unchanged over time (Pearman et al., 2008). Under this assumption, a species future distribution can be predicted according to climate changes based on their historic niche conditions. **Chapter 6** contests this assumption, providing evidence that several Odonatan species have not exhibited a consistent realised climate niche on a national scale during a recent period of climate change. It is therefore exceedingly difficult to predict the implication of climate change for biodiversity as species will not necessarily track climate changes. Niche flexibility offers a novel and understudied means through which species may face climate change; damselflies (Zygoptera) which were more able to tolerate different climate conditions during a period of climate change were found to be less at risk in terms of loss of range size (**chapter 6**). Therefore, the ability of species to shift their climatic niche instead of geographically tracking their historic climate conditions is likely to be an important mechanism of species survival under continued climate change, particularly for species which are not able to keep pace with climate changes.

Geographic shifts are an important means through which species can persist during climate change (Wallingford et al., 2020), with measures to increase habitat connectivity and facilitate species movements considered crucial aspects of biodiversity management. This research, however, highlights the multitude of ecological impacts through which species on the move with climate change can exert on other native species within recipient environments (**chapter 7**); species distribution changes have the potential to alter biological interactions, disrupting ecosystems and affecting the societies that reply on them (Pecl et al., 2017; Twiname et al., 2020). **Chapter 7** demonstrates a novel implementation of invasive frameworks for species that are shifting due to climate change as a readily available, thorough, and transparent way of considering potential ecological impacts. This research demonstrates specific negative and positive impacts of the Odonata and Hymenoptera (wasps, bees, ants, and sawflies) study species on other UK natives and transposes this evidence into a spatial, semi-quantitative assessment for each taxonomic group through a projection of species' future distributions under projected climate change (**chapter 7**). Thereby this chapter demonstrates concrete and specific examples of potential ecological implications of species on the move gathered from the scientific literature, as well as illustrates the applicability of invasive frameworks for climate-tracking species for the first time.

8.2. Limitations of approach

This section outlines the main limitations of the methodology employed in this work. Firstly, this research being geographically restricted to the UK, necessitates a consideration of the extent to which these findings are generalisable to other localities as well as a wider examination of the constraints to national scale investigations. The specific responses by species inspected here are according to the distinct climate system of the UK due to its unique combination of size, location within the mid-latitudes, proximity to the Atlantic Ocean and Gulf Stream, and the influence of the prevailing southwesterly winds. Likewise, each species response is mediated by its particular distribution within the UK with distinctions between widespread, southerly, and northerly distributed species which are adapted to different climatic conditions. Therefore, though it might be surmised that conclusions regarding species response to climate change would be transferrable to other temperate, mid-latitude localities, the distinct climatic features of the UK should also be recognised. Nevertheless, the principal findings of this thesis are broadly generalisable, for example, there is no rationale for presuming that species occurrence records would not likewise vary between sources in other localities (**Chapter 5**), or that in other countries certain species would not similarly exhibit a temporal divergence in their realised climate niche (**Chapter 6**). Indeed, biological records from social media have been previously found to complement traditional recording in other locations such as for honeybees in Australia (ElQadi et al., 2017), baboon spiders in Southern Africa (Campbell and Engelbrecht, 2018), and the Eurasian red squirrel in Portugal (Rocha et al., 2017), and niche shifts during climate change have been previously observed for bird species in both North America (Ralston et al., 2017) and Italy (Tirozzi et al., 2022).

The confinement of this research to the UK rather than across our UK native species' entire geographic range requires further scrutiny beyond the generalisability of findings as it influences the validity of the conclusions drawn across all data chapters. **Chapter 5** establishes that distinct data types capture different aspects of the Banded Demoiselle's (*C. splendens*) environmental niche, however, we cannot make any conclusive determination as to whether this would be the case if comparing different data sources across the Banded Demoiselle's (*C. splendens*) entire geographic area. The results of **chapter 6** indicate that even when only including analogue climate conditions between two time periods, species exhibit a temporal divergence in their realised climatic niche, yet it remains unclear whether species are perhaps occupying the same climatic conditions over time at other localities outside the UK within their wider distribution. Finally, investigation of the spatial dispersion of potential environmental impacts of species redistribution under a future climate scenario for **chapter 7**, suggests that impacts will be lowest in south UK, whereas it is these areas that might be theorised to be most at risk under climate change due to the arrival of new species from the continent along with accompanying novel biotic interactions. Despite this, the conclusions drawn remain informative as many policies operate on national scales. To offer an example, the temporal niche divergence of species observed in **chapter 6** undermine the assumptions of SDMs if employed on similar national scales – as is often the case (see for example Razgour et al., 2011; Giannini et al., 2012; Oliver et al., 2012; Rapacciuolo et al., 2014; Geary & Von Hardenberg, 2020; Freeman et al., 2022; Bourhis et al., 2023; Rodríguez-Roy et al., 2023) – which might overwise be mitigated on larger spatial scales when considering a species entire geographic distribution.

A further consideration in terms of the applicability of the findings presented in this thesis in broader contexts, is the restriction of the analysis to two mobile taxa within the Insecta class, Odonata (**chapters 6** and **7**) and Hymenoptera (**chapter 7**), and to a single Odonate species – the Banded Demoiselle (*C. splendens*; **chapter 5**). The results presented cannot be readily generalised across different species, invertebrates, or even other Insecta taxa; particularly as **chapter 6** indicated distinct differences between dragonflies and damselflies, which are closely related species that might by hypothesised to respond similarly to climate change despite their morphological differences. To illustrate, damselflies were shown to exhibit a negative correlation between increased occupancy and niche overlap (**chapter 6**), consistent with previous research for birds in Northern Italy (Tirozzi et al., 2022), yet this relationship was not observed for dragonflies, nor did the relationship hold when all Odonata species were included. The distinction between these findings for dragonflies and damselflies were supposed to be a result of differences in dispersal ability, therefore the response of other species to climate change might likewise be theorised to be influenced by dispersal. Less mobile, non-flying Insecta might be less apt at tracking climate change, for example. Regardless of the distinctive correlations detected between species groups, the principal finding of **chapter 6** – that temporal niche conservatism cannot be assumed for all species on a national scale – remains generalisable to other contexts, with niche divergence detected within both dragonflies and damselflies.

For the investigation into the applicability of social media data (**chapter 5**), the Banded Demoiselle (*C. splendens*) was chosen specifically because it has an interesting appearance and is particularly easy to identify. Less charismatic species which are more difficult to identify by non-experts, will have a smaller availability of social media records, with observations also more difficult to extract if the species is unidentified and the name is not included, and the process of verification after data collection might be significantly more difficult and time-consuming. Moreover, it is difficult to extrapolate these results to other localities, as engagement with social media, even for charismatic species, might not be as considerable elsewhere. In fact, the most remote and understudied areas in terms of biological recording might lack the infrastructure and connectivity for social media to be a feasible option. This case study does nonetheless provide a compelling case for the advantage of social media data and the variability between different sources of biological records (**chapter 5**). Furthermore, lacking a standard protocol for the collection of species distribution data from social media applications (Chowdhury et al., 2024), this research demonstrates an example methodology for the collection and verification of records which can be applied to other species in other contexts.

A further consideration of the research presented in this thesis, is that all the analyses were based on presence-only, opportunistic, and largely unstructured species occurrence records. Other biological recording, such as abundance records, structured surveys, or atlas data have the advantage of standardised methods and comprehensive spatial coverage. Moreover, abundance information provides insight into population dynamics which can provide a more accurate picture of species' distribution, habitat suitability, and response to environmental change. Whereas the biases associated with opportunistic observations can result in skewed representations of species distributions and distorted trends when certain areas or periods are overrepresented. Additionally, the lack of information on species' absence in presence-only data complicates the estimation of a species genuine distribution as well as hinders the ability to accurately model the variables that influence species presence and absence. Even when there is an attempt to identify all species present at a given location, or to sample species presence and absence over a given area, detection is imperfect with false absences common (Isaac & Pocock, 2015). Unstructured records are further biased by uneven detectability, for example according to the accessibility of a given locality, varying time of year, and observer experience and skill. The vast majority of the Odonata observations included are of flying adults (**chapters 5, 6** and **7**) due to an improved detectability in comparison to the larval stage, however, there is large debate regarding the suitability of such records as given that adult Odonates are highly mobile, their presence does not necessarily indicate the suitability to support a viable population (Pearce-Higgins & Chandler, 2020). Adult occurrences have been found to overestimate habitat suitability, particularly for the most mobile and more generalist species (Raebel et al., 2010). Therefore, there is a degree of uncertainty to the extent to which this investigation adequately captures the true distribution and habitat of the Odonate study species. Exuviae occurrences capture proof of life-cycle completion at a site (Raebel et al., 2010), however, at the expense of lower detectability and risk of underestimation of occurrence (Giugliano et al., 2012; Pearce-Higgins & Chandler, 2020). Abundance records would be more informative, with counts being shown to correlate with sampling of the larval stage that more adequately captures species aquatic niche requirements (Pearce-Higgins & Chandler, 2020). While acknowledging the inherent limitations of the species data utilized in this thesis, it is important to recognize that this occurrence data was selected for its extensive availability and breadth of coverage. Substantial abundance records for the chosen taxa are not currently available in the UK and would require vast number of volunteers with accessible methods and adequate species identification expertise (Pearce-Higgins & Chandler, 2020). Furthermore, methodological developments to address issues of bias and model species from presence-only records enable an accurate implementation of this type of occurrence records; various techniques have been explored throughout this thesis, for example, the implementation of pseudo-absences and bias covariate correction methods for modelling habitat suitability (**chapters 5** and **7**; Chauvier et al., 2021; Warton et al., 2013), extracting species trends using the Telfer Index (**chapter 7**; Telfer et al., 2002), and employing local benchmark species through the Frescalo (FREquency SCAling LOcal) method to filter data and account for spatial bias (**chapter 6**; Hill, 2012).

Chapter 5 provides a comparison between occurrence types, separated into traditional and citizen science, however, these categories are not wholly distinct from one another with both types offering unstructured, opportunistic records (Isaac et al., 2014). If available, it would have been preferable to compare unstructured biological records and social media observations with abundance records or observations from structured monitoring schemes. For the purpose of this study, biological records labelled 'traditional' were primarily from the British Dragonfly Society (BDS) and those directly gathered from iRecord or iNaturalist were labelled as 'citizen science', presuming that records directly from the BDS would have been collected and verified with by Country Recorders with a level of expertise and greater degree of reliability and accuracy. There is nonetheless a significant degree of overlap between these record types and therefore uncertainty regarding how distinct these categories are. As records from the BDS are fed directly into iRecord, records labelled as such were removed from the citizen science database, however it is also likely that several members of the BDS frequently record sightings through iRecord as well as directly to the BDS. Although this separation is not complete and occurrences may be present in both datasets, differences were nevertheless detected and any meaningful conclusions drawn regarding the additional information that can be garnered from social media remains relevant (**chapter 5**).

There is further uncertainty regarding the social media records employed which lacking geo-tags or precise location information could not be confined to a single 1-km grid on which the SDMs were implemented (**chapter 5**). Instead, each grid square covering the broader area in which the Banded Demoiselle was recorded was included as a presence; this pseudo-replication could potentially lead to an overestimation of the Banded Demoiselle's range and habitat preference as well as potentially inflating recorder bias, both affecting the apparent distinction from traditional and citizen science occurrences (**chapter 5**). In this case sensitivity testing filtering data to varying levels of precision provided little evidence that the low precision of social media records significantly affected results, therefore all available observations extracted from social media were included to provide a full investigation (**chapter 5**). This is unlikely to be the case in all instances therefore this methodology should be employed with care and comparable evaluation of precision. Another aspect to consider for the social media records is the potential for increased recorder bias, resulting in uncertainty regarding whether the increased representation of data from urban areas is a true one or an artifact of that bias. Distance to nearest roads was an important predictor for models based on social media observations but not for citizen science and traditional records (**chapter 5**). A bias covariate correction method was employed to account for this recorder bias (**chapter 5**; Warton et al., 2013), with corrected models differing around major population centres. There is uncertainty, however, in the extent to which the single recorder effort proxy adequately captures and accounts for varying sampling effort, particularly as the other chosen proxies did not significantly affect models. A target-background approach is a widely implemented alternative to account for recorder effort in the implementation of SDM (Phillips et al., 2009), however this was not feasible for the social media dataset. Despite these limitations, this research provides a useful appraisal of social media records as well as a method for extraction of biological records from social media platforms and implementation for SDM, including an exploration of the impact of various levels of spatial precision and consideration of reliability.

To determine temporal changes in species climate niche and distribution a simple methodology was employed comparing these aspects in two discrete time periods (**chapter 6**). A limitation to this methodology is that non-linear effects might have been missed; although this method reveals that several species climate niche differed during a recent period from that in the past, a time series analysis at a higher temporal resolution would have been more informative though data intensive. Simply including the intermediary period for which occurrence data was collated to reveal species occupancy trends, however, would have been advantageous and was only neglected for simplicity and brevity of the investigation (**chapter 6**). Moreover, the geographic distribution change of species was determined through a simple centroid change analysis capable of revealing often neglected multidirectional changes (Lenoir & Svenning, 2015), with the advantage of being appropriate for rapid computation over several species (**chapter 6**). This methodology, however, overlooks several aspects of species' range and population dynamics, without including information on changes to species' range margin or abundance, previously revealed to be mediated by climate change (Bowler et al., 2015). This technique is also sensitive to outliers that can skew results, as well as being inappropriate for species with widespread, disjunct or fragmented distributions. With the availability of abundance records more nuanced measures are possible such as change in population centroid (Liang et al., 2021) and abundance-weighted centroid (Ash et al., 2014). For presence-only data, a valuable alternative is presented by Gillings et al. (2015) who measured multidirectional range shifts for British birds, focussing on distributional changes at species' leading-edge. Changes in species mean elevation were also calculated and mean changes compared between dragonflies and damselflies (**chapter 6**), however, due to the high variability between species with some species increasing and others decreasing their elevation, mean changes across these groups were small and not biologically meaningful, obscuring more complex patterns.

Chapters 5 and **7** of this thesis both apply correlative SDMs, a data driven methodology to link species occurrence data to environmental data. Similarly, the temporal niche analyses applied in **chapters 6** and **7** are based on an ordination approach that investigates the association between species occurrence information and climate data. Therefore, the techniques employed here are founded on a correlative approach, without any explicit modelling of the underlying biological processes and mechanisms which drive species' response to climate change. This approach can obscure results, for example, investigation of the correlation between species' occupancy and niche dynamics revealed different statistically significant relationships for damselflies and dragonflies (**chapter 6**), however, due to the low sample size for each group as well as the high variability between species, it can be debated to what extent these differences reveal a true distinction between these subgroups. Other variation between species might offer a more convincing explanation of the differences detected, for example, differences between species that exhibit high retention versus those with a less stable distribution. Although biological differences between dragonflies and damselflies offer a compelling explanation of the findings presented (**chapter 6**), further investigation is required to confirm these inferences. Employing physiological data and experimental approaches enhances mechanistic understanding and can improve model predictions and explanatory power, however, such techniques are data and time intensive.

The underlying mechanism relating temperature to Odonate distribution is an important aspect that is overlooked in this study that requires further work (Pearce-Higgins & Chandler, 2020). It is difficult to attribute occupancy and distribution changes to climate change as ambient conditions have been shown to significantly influence the detectability of several Odonate species with individual activity and behaviour being positively related to warmer conditions (Pearce-Higgins & Chandler, 2020). Moreover, it is likely that recorder effort is also biased by weather variables, with greater sampling effort expected during more favourable weather conditions. Pooling species observations into two extensive time periods to investigate the influence of long-term climatic changes can to an extent mitigate these influences by averaging out short term weather fluctuations, however, there remains valuable insight to be gained from further mechanistic examinations. Another significant factor that is overlooked is the potential influence of freshwater processes on Odonata distributions (**chapter 6**). This constitutes a considerable limitation of this work as there is evidence that on a family-level, recent distribution changes that are often attributed to climate change, such as the northward range expansion of freshwater invertebrates in Britain are not correlated with temperature but in fact better explained by significant improvements in water quality in Northern England (Vaughan & Ormerod, 2014). Accordingly, it is likely that the distribution and occupancy changes detected here for Odonates (**chapter 6**) are related or at least mediated by trends in water quality which have coincided with recent climatic variability (Vaughan & Ormerod, 2014). Such changes as well as other freshwater processes might likewise account for some of the differences detected between niche tracking and niche flexible species (**chapter 6**). Although this research goes beyond a simple unidirectional temperature-centric approach, it suffers another common shortfall in species redistribution studies whereby alternative non-climatic causal factors are not sufficiently investigated (Taheri et al., 2021). This study would be improved by a sufficient investigation into the potential influence of water quality, as well as other factors such as land-use or habitat disturbance on the detected distributional trends of species. Despite this, the correlative approach employed here, although not directly interrogating the specific processes through which climate changes affects species, allows for a broad investigation across numerous species that makes use of readily available data and reveals overarching correlative relationships, such as the increase in occurrence of damselflies with a greater niche flexibility (**chapter 6**).

A final limitation to consider is the assumed parallel between species on the move with climate change and invasive species, on which **chapter 7** is established. Although similarities have been established (Pettorelli et al., 2019; Wallingford et al., 2020), there are distinct differences between species which have been introduced through human activities and those which are shifting into new areas by tracking climate change (see Urban, 2020 for a summary of these distinctions). As such, there are limitations to employing an invasive framework to assess the impacts of climate-tracking species; the primary discrepancy which leads to complications is the degree of exposure to novel environments and communities. The potential impacts detected were primarily relating to biotic interactions between species and the degree of impact determined based on the measured level of change induced on other native species in the context of the literature assessed. It is difficult to equate these findings to actual impacts for climate-tracking species without any assessment of whether these biotic interactions will occur as species shift and how novel they will be. Nevertheless, **chapter 7** demonstrates for the first time that it is possible to implement an invasive framework for climate-tracking species and even if potential impacts require further investigation this methodology provides a thorough and structured means of undertaking an initial assessment.

Overall, despite several limitations in the approach taken, this research makes several key advances to the study of species redistribution science. Although this study was undertaken on a national scale for the UK with two specific taxonomic groups the principal findings are broadly generalisable to other contexts and the methodologies employed provide a standard through which to assess other species in other localities. Moreover, employing presence-only occurrence records through a data-driven correlative approach allowed for broad multidirectional distribution changes to be assessed in relation to current and future climate change, and revealed novel relationships across several species.

8.3. Recommendations for further research

This research has developed novel methods for investigating species on the move with climate change which advance species redistribution science; these techniques can be developed and applied to additional species and localities under other contexts to further contribute to the field. Moreover, the advancements made in this research accentuate remaining knowledge gaps and challenges within this discipline. Therefore, this section provides directions for future research to better understand species movements under climate change and their ecological implications in order to support biodiversity conservation.

Integrating a variety of biological records provides a more comprehensive impression of a species geographic distribution and environmental niche (**chapter 5**), therefore, further work is required to broaden the scope of novel data types such as the development of techniques for automated retrieval and verification of social media data (ElQadi et al., 2017). Continued implementation of occurrence data from citizen science platforms is also important and an indispensable means of public engagement; further development of such platforms for inputting information regarding recorder effort and utilisation of GPS location data, would greatly enhance biological recording efforts and reduce errors relating to the ‘recorder effort problem’ and incorrect georeferencing (Yesson et al., 2007; Isaac & Pocock, 2015). It would also be valuable to combine unstructured observations with more intensive, structured surveys and abundance information, for example, to provide insight into population dynamics at species range margins under environmental change and the aspects influencing species habitat suitability. Moreover, having demonstrated the benefit of social media for biological recording (**chapter 5**), it would be useful to trial this methodology for other species in other areas, particularly in cases where traditional observations are lacking (Allain, 2019). The approach employed here gathered information from existing historic social media content, however, a more active and ongoing approach offers an interesting area for future study; for example, by setting up a specific Facebook page or hashtag on Instagram or Twitter to encourage submission of observations of a target species or taxa.

This thesis draws attention to a consequential research question – do species track climate change? It has been largely assumed to date within species redistribution science that species typically conserve their environmental niche over time and as such track climate changes geographically in order to remain within optimal, historic climate conditions (Scheffers et al., 2016). Accordingly, much attention within this discipline has been assigned to describing species climate niche to predict how species might respond to climate changes (Guisan et al., 2014), such as through the implementation of SDMs (Willis et al., 2015) or calculation of the local climate velocity of a species (Burrows et al., 2011). Here it is revealed that species do not necessarily conserve their climate niche over time on a national scale during climate change (**chapter 6**). This finding necessitates further research into the extent to which species can track climate changes geographically and alternative responses under environmental change. Integration of morphological, physiological, and behavioural data would enable further exploration into the specific factors which mediate species response, such as those contributing to a species ability to track climate changes or shift their environmental niche. Underpinning the mechanistic processes would improve our ability to predict species response and could elucidate the varied outcomes for different species and different taxonomic groups (such as the distinctions between dragonflies and damselflies – **chapter 6**). Furthermore, comparing the temporal variance in species realised climate niche across species' entire geographic range would provide critical information regarding niche conservatism and the influence and utilisation of non-analogue climates – such as previous climate habitat lost under climate change as well as novel climate conditions.

The potential for adverse as well as beneficial impacts of species on the move with climate change for other native taxa has been affirmed by this research (**chapter 7**). Despite the limitations and concerns with drawing a parallel between invasive and climate-tracking species (Urban, 2020), it is evident that structures need to be in place to anticipate and measure the potential ecological consequence of species distribution change (Pettorelli et al., 2019). Having been widely applied to assess the impact of species arrivals in new environments, invasive frameworks offer a valuable place to start. The applicability of the Environmental Impact Classification for Alien Taxa (EICAT) and EICAT+ frameworks for assessing the environmental impacts of climate-tracking species has been demonstrated (**chapter 7**); it would be beneficial to also explore the application of the Socio-Economic Impact Classification of Alien Taxa (SEICAT; Bacher et al., 2017) as well as developing a SEICAT+ to evaluate adverse and beneficial effects, respectively. Combining methodologies implemented throughout this research presents an effective approach to fully evaluate the consequence of species movements by weighing anticipated impacts through biotic interactions revealed by invasive assessments (**chapter 7**) according to the degree of novelty of these interactions. For example, assessing the level of coexistence between specific species under current and future scenarios within a given locality by predicting species' changing habitat suitability with SDMs (**chapter 7**) and evaluating changing niche overlap between species (**chapter 6**). It is theorised that the adverse effects of a climate-tracking species on another native taxa would be more severe if previous coexistence geographically and in terms of species' environmental niche is lower due to a greater degree of novelty in species biotic interactions – testing is necessary to corroborate this hypothesis.

8.4. Conservation challenges and implications

Range shifting species pose a dilemma for conservation, as climate change can threaten species within their historic range, whilst the same species could adversely harm biodiversity within their new localities (Cranston et al., 2023). The success of conservation management and restoration approaches are often measured using a baseline or reference state (McNellie et al., 2020), however, climates are moving into states without any present-day or historic analogue (Arneth et al., 2020; Conradi et al., 2024), species movements are forming new interactions (Pecl et al., 2017) and subsequently novel ecosystems are produced (Ordonez et al., 2024) and biomes may even change (Huntley et al., 2021); measuring progress towards achieving 2030 biodiversity targets such as restoring degraded ecosystems may be hindered by such potential no-analogue conditions.

The operation of Protected Areas (PAs) within biodiversity conservation is a contentious issue that has been widely critiqued in the wake of climate change. For example, a lack of climate connectivity may inhibit species movements through the PA network (Parks et al., 2023), undermining their effectiveness (Farooqi et al., 2022). Despite criticisms, expanding the global network of PAs remains an important solution to the biodiversity crisis (Dobrowski et al., 2021; Parks et al., 2022), therefore it will be crucial to consider climate-driven changes when delineating new PAs (Dobrowski et al., 2021). For example, employing insights into species changing distributions (**chapter 7**; Dobrowski et al., 2021) and consideration of future climate analogues to strategically expand the PA network to improve climate connectivity and reduce the prevalence of novel and disappearing climates (**chapter 6**, Parks et al., 2022). Furthermore, as species movements are difficult to predict (**chapter 6**) and novel communities will emerge due to changing biotic interactions (**chapter 7**), species monitoring within the PA network will also be crucial to build understanding of how ecosystems are changing with climate change to adapt management practices where required.

Species which are unable to keep pace with climate change by shifting geographically, nor having the niche flexibility to adapt to changing climates, might be particularly vulnerable and likely to undergo declines (**chapter 6**). This was demonstrated for damselflies, where a lower niche flexibility during climate change was correlated with greater distribution declines, theorised to be due to damselflies limited dispersal capacity compared to dragonflies. Under such pressures, some scientists have advocated new conservation measures such as the managed relocation of species (Cranston et al., 2022; Parks et al., 2023) – the movement of individuals of a threatened population into new areas where the species would be predicted to shift to without the constraints of habitat availability, climate connectivity, anthropogenic barriers, or dispersal ability (Twardek et al., 2023). Although aimed to facilitate species survival, such strategies will require thorough consideration of potential environmental and socio-economic outcomes resulting from novel biotic interactions and changing community dynamics (**chapter 7**).

The need for a transparent and value-three approach to conservation was highlighted in **chapter 7** of this thesis. Whilst invasive species are accepted as one of the key drivers of biodiversity loss (Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services [IPBES], 2019) with targets to reduce the introduction of invasive alien species by 50% by 2030 (CBD, 2022), there are instances where alien species can positively influence native taxa and biodiversity (Vimercati et al., 2022). Likewise, climate-shifting species are often considered as an unmitigated good without consideration of potential negative impacts (Twiname et al., 2020). Application of impact assessments to review positive and negative consequences of new arrivals, either anthropogenically introduced or driven by climate change, offer a system to measure potential impact severity (**chapter 7**), though these assessments cannot inform management actions alone (International Union for the Conservation of Nature [IUCN], 2020b). A focus on the conservation of ecosystem functioning offers a potential solution, which could be secured by native, climate-tracking, or even alien species (Hermoso et al., 2022). Development and implementation of a consistent set of metrics to measure functional diversity and ecosystem functioning would therefore be beneficial in conjunction with continued species monitoring efforts.

Public and expert opinion will influence management actions under climate change. How species are viewed might help or hinder conservation efforts, such as species perceived as threatened or charismatic compared to harmful or negative (Cranston et al., 2022). Engagement with wildlife recorders offers a potential means to gather support for conservation actions of species under climate change (Cranston et al., 2022). Social media and citizen science, as well as more fully capturing species geographic distribution and environmental niche (**chapter 5**), offer a significant opportunity to engage with the wider public and influence opinion.

A further challenge for conservation under climate change is the uncertainty in species' response, as species are not equally capable of tracking changes geographically and vary in the degree of consistency in their climate niche (**chapter 6**). There are also uncertainties in SDM projections under climate change, especially if assumptions of niche conservatism do not hold (**chapter 6**) and when outputs vary between types of biological records (**chapter 5**). To deal with uncertainty within climate change research, modelling future climate dynamics are based on a suite of plausible scenarios; anticipating potential biodiversity scenarios might likewise be a valuable management tool (Hermoso et al., 2022; Farooqi et al., 2022). For example, by prioritising conservation investments in areas predicted to be consistently important for a suite of species under different timescales and different uncertainty levels (Carvalho et al., 2011). Building ecosystem resilience will also be important to conserve biodiversity and mitigate climate change impacts (Farooqi et al., 2022) recognised in the 2030 biodiversity targets (CBD, 2022). This could involve enhancing habitat connectivity to facilitate species movements and recovering disturbed environments (Farooqi et al., 2022). Another target is to enhance green spaces and urban planning for joint human and biodiversity benefits (CBD, 2022), this is likely to further facilitate species movements, particularly as the importance of urban habitat may be underestimated by traditional biological records (**chapter 5**).

9. Conclusion

This thesis furthers the field of redistribution ecology and improves our ability to measure, predict, and anticipate the implications of species on the move with climate change. The response of species to climate change are evaluated, focussing on Odonata (dragonflies and damselflies) and Hymenoptera (wasps, bees, ants, and sawflies). This includes an investigation of species' multidirectional geographic distribution shifts that utilises a suite of average and extreme precipitation and temperature variables. Ordination methods are employed to determine the significance of these climate variables and measure the degree of consistency in species' realised climate niche over time. Simultaneously analysing species' geographic and niche shifts during recent climate change allows for a novel assessment of the correlation between species shifts and the temporal consistency in their occupied climate conditions. These methods indicated an interesting variability between dragonflies' (Anisoptera) and damselflies' (Zygoptera); whilst the most apt climate-tracking dragonflies were able to shift to keep pace with climate changes, damselflies were less vulnerable to climate change when exhibiting a degree of flexibility in their realised climate niche. Such findings emphasise that not all species are tracking their historic climate conditions with significant ecological implications for national scale niche models.

This research also demonstrates a methodology for extracting species occurrence data from social media applications which provide valuable insights into species geographic distributions and habitat requirements. Such data is complementary to traditional and citizen science information sources, therefore, there is value to be gained from integrating diverse data types. With continued technological advancements and growing use of social media and mobile phones, such data is likely to become increasingly significant for ecological studies. Improvement of automated verification techniques for biological recording is a key scientific priority for future work.

As it is known that species shifts with climate change will have several environmental and societal consequences, by focussing on the arrival of climate-tracking species in new localities, a spatial assessment can be employed to predict potential impacts for other native species. This is demonstrated here by modelling species' potential future habitat suitability and employing an impact scoring framework based on insight from invasion biology. As such, the practical application of invasion frameworks for species on the move with climate change is evidenced here for the first time.

The findings presented within this thesis can influence future species' management and conservation. Future study should take full advantage of the range of biological information available, further examine temporal niche conservatism during past climate change, as well as continue to advance frameworks to evaluate and predict the impact of species redistribution.

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Appendices

Appendix A: Banded Demoiselle occurrences from social media

Occurrences of the Banded Demoiselle (Calopteryx splendens) in Great Britain from 2010 onwards were sourced from social media applications including Twitter (table A.1), Facebook (table A.2), and Flickr (table A.3). Occurrences were manually sourced from Twitter and Facebook and extracted from Flickr using the Flickr API flickr.photos.search using the search terms ‘Banded Demoiselle’ and ‘Calopteryx splendens’. Occurrences consisted of an image or video to confirm correct species identification (through manual verification), location information, and date of sighting, or if unavailable the date the content was posted. The occurrences presented in these tables include the date the Banded Demoiselle was recorded when available, the date the social media content was posted, the location description given, the latitude and longitude (this is provided through the Flickr search, for Facebook and Twitter this was extracted as the centre point of the given location), and the web address for the social media content. As more precise geo-location information for observations from Twitter and Facebook were not provided, these also include a notation of all the 10-km and 1-km British National Grid cells which cover the entire location description provided.

Table A.1. Banded Demoiselle (Calopteryx splendens) occurrences manually extracted from Twitter.

Date Recorded	Date Posted	Location Description	Latitude	Longitude	10km Grid	1km Grid	Website
06/09/2020	06/09/2020	Morden Hall Park, London, Morden, SM4 5JD	51.40283	-0.18667	TQ26	TQ2668 TQ2669 TQ2568 TQ2569	https://twitter.com/ArjunDutta230/status/1302659367393787904
03/08/2021	03/08/2021	Glascoed Nature Reserve, Bodelwyddan, Denbighshire	53.2545	-3.48194	SJ07	SJ0174	https://twitter.com/marcbuzzard/status/1422596650514661377
Not Provided	13/07/2021	Portland Bird Observatory	50.51968	-2.45109	SY66	SY6868	https://twitter.com/PortlandBirdObs/status/1414928337357656069
Not Provided	06/07/2021	Roughmoor Pond, Taunton	51.02288	-3.12331	ST22	ST2125	https://twitter.com/ConnollyJohng/status/1412381186916290565
09/06/2021	09/06/2021	Watermead Lake	51.83351	-0.80835	SP81	SP8215	https://twitter.com/DanFWildlife/status/1402639573059084292
08/06/2021	10/06/2021	Great Amwell	51.79423	-0.01368	TL31	TL3613 TL3713 TL3612 TL3712 TL3611	https://twitter.com/wibsdad/status/1402900799588605953

10/06/2021	10/06/2021	RHS Garden Wisley	51.31248	-0.47353	TQ05	TQ0658 TQ0657 TQ0659	https://twitter.com/I_Cavadino/status/1402960575278333958
11/06/2021	11/06/2021	Wilford Claypit Nature Reserve	52.91417	-1.15298	SK53	SK5735 SK5636	https://twitter.com/SnowShoe_Sue/status/1403334472695746565
Not Provided	19/06/2021	Dosthill Park	52.60419	-1.68636	SK20	SK2100	https://twitter.com/gavinpordage/status/1406265461889765386
Not Provided	23/06/2021	River Otter, Ottery Saint Mary	50.75056	-3.28539	SY09	SY0995	https://twitter.com/OtteryBirder/status/1407725773369323522
23/06/2021	24/06/2021	River Wye, Dixton	51.8182	-2.69748	SO51	SO5213	https://twitter.com/MikeCram5/status/1407941160564310016
05/06/2021	05/06/2021	River Severn, near Highley	52.44884	-2.36556	SO78	SO7484 SO7483 SO7583	https://twitter.com/MarkRog70739206/status/1401268697893937155
05/06/2021	05/06/2021	Measham	52.706325	-1.506681	SK31	SK3212 SK3312 SK3211 SK3311	https://twitter.com/smithymik/status/1401189048027430912
Not Provided	01/06/2021	RSPB St. Aidan's Nature Park	53.75111	-1.40863	SE32	SE3829 SE3929 SE3828 SE3928 SE3827 SE3927	https://twitter.com/marcbuzzard/status/1399616402991030273
Not Provided	28/07/2020	Hankridge Water Park, Taunton	51.02528	-3.06866	ST22	ST2525	https://twitter.com/JohnTolfree/status/1288190713390411777
Not Provided	13/01/2022	Walthamstow Wetlands	51.58564	-0.04909	TQ38 TQ39	TQ3490 TQ3590 TQ3489 TQ3589 TQ3488 TQ3588 TQ3488	https://twitter.com/wilder_action/status/1481601907022049282
Not Provided	23/06/2020	Grimston	52.77425	0.5448	TF72	TF7122 TF7222	https://twitter.com/_TCartz/status/1275382431563100160
Not Provided	21/06/2020	Elkesley	53.27289	-0.97017	SK67	SK6875 SK6975	https://twitter.com/stevewilko11/status/1274723867735535619

Not Provided	21/06/2020	Lea Valley Country Park	51.71423	-0.01321	TL30	TL3605 TL3604 TL3603 TL3602 TL3601 TL3701 TL3702 TL3703 TL3704 TL3505	https://twitter.com/Greenwoodworker/status/1274666771316957184
14/06/2020	14/06/2020	Upper Poppleton	53.979873	-1.152983	SE55	SE5553 SE5554	https://twitter.com/Sandham_C/status/1272170125425328130
Not Provided	09/06/2020	Forest Farm, Cardiff	51.51959	-3.2488	ST18	ST1380	https://twitter.com/stubbingtonbear/status/1270420936882368516
01/06/2020	05/06/2020	Park Calon Lan, Blaengarw	51.62528	-3.59099	SS89	SS8993 SS8992	https://twitter.com/wandering_rjt/status/1269015880429641728
02/06/2020	03/06/2020	Red squirrel trail, Shanklin	50.63347	-1.17073	SZ58	SZ5882 SZ5881	https://twitter.com/mdg219/status/1268090933075156992
Not Provided	02/06/2020	River Tone, Taunton	51.01655	-3.10897	ST22	ST2125 ST2224 ST2225 ST2325	https://twitter.com/ejwwest/status/1263866648504786951
Not Provided	22/05/2020	Fishlake Meadows	51.00392	-1.49248	SU32	SU3522 SU3523	https://twitter.com/Lukehepps79/status/1254166195042910209
Not Provided	25/04/2020	Trowbridge	51.32011	-2.20852	ST85	ST8456 ST8457 ST8556 ST8557 ST8656 ST8657 ST8458 ST8558 ST8658	https://twitter.com/Lukehepps79/status/1254166195042910209
Not Provided	24/07/2019	Watermead, Aylesbury	51.83526	-0.80936	SP81	SP8215 SP8216 SP8115 SP8116	https://twitter.com/WatermeadNature/status/1174819467177668608

Not Provided	30/08/2019	Moors valley, Hampshire	50.85412	-1.83425	SU10	SU1005 SU1105 SU1205 SU1006 SU1106 SU1206 SU1107 SU1007	https://twitter.com/WatermeadNature/status/1174819467177668608
21/07/2019	21/07/2019	St Asaph	53.25782	-3.44636	SJ07	SJ0374	https://twitter.com/marcbuzzard/status/1152864474115842048
Not Provided	20/07/2019	Potteric Carr	53.4939	-1.10084	SE50 SE60 SK59 SK69	SE5800 SK5899 SE5900 SK5999 SE6000 SK6099	https://twitter.com/ancienprof/status/1152509455793381376
Not Provided	17/07/2019	Llangaffo	53.1901	-4.33024	SH46	SH4468	https://twitter.com/cathy_bug/status/1151580059884490752
Not Provided	08/07/2019	North Hinksey	51.74888	-1.29636	SP40	SP4806 SP4805 SP4905 SP4904 SP4804	https://twitter.com/oldgreyheron/status/1148303554513444864
Not Provided	02/07/2019	Attingham Park	52.68673	-2.6717	SJ51 SJ50	SJ5410 SJ5510 SJ5509	https://twitter.com/Naturedoorstep/status/1146126618835992577
Not Provided	29/06/2019	Abergavenny	51.82927	-3.01728	SO21 SO31	SO2914 SO2915 SO2814 SO2815 SO3015 SO3014 SO3013	https://twitter.com/Openfoot/status/1144945771399323648
23/06/2019	23/06/2019	Heather Farm, Horsell common	51.34018	-0.56673	SU96	SU9960 SU9961 TQ0060 TQ0061	https://twitter.com/MiradorDesign/status/1142920674316562433
18/06/2019	19/06/2019	Heather Farm, Horsell common	51.34018	-0.56673	SU96	SU9960 SU9961 TQ0060 TQ0061	https://twitter.com/MiradorDesign/status/1141265744350629888
Not Provided	17/06/2019	Brandon Marsh, Coventry	52.37519	-1.43219	SP37	SP3875 SP3975	https://twitter.com/WsSeaton/status/1140606630339502080

06/06/2019	06/06/2019	Glassonby	54.74208	-2.66201	NY53	NY5738 NY5739 NY5638 NY5639	https://twitter.com/EdenRiversTrust/status/1136632291684339715
01/06/2019	01/06/2019	Heather Farm, Horsell common	51.34018	-0.56673	SU96	SU9960 SU9961 TQ0060 TQ0061	https://twitter.com/MiradorDesign/status/1134956527582949382
Not Provided	02/06/2019	Walthamstow Wetlands	51.58564	-0.04909	TQ38 TQ39	TQ3490 TQ3590 TQ3489 TQ3589 TQ3488 TQ3588 TQ3488	https://twitter.com/Phil3Production/status/1135253695409532928
01/06/2019	01/06/2019	Mildenhall	52.34464	0.51021	TL77	TL7175 TL7174 TL7074 TL7075	https://twitter.com/birdingprof/status/1134855250333110274
Not Provided	20/05/2019	Bulkington	52.4772	-1.42637	SP38	SP3986 SP3987 SP3886 SP3887	https://twitter.com/TomEdwardsWild/status/1130534614844497920
18/05/2019	19/05/2019	Bulkington	52.4772	-1.42637	SP38	SP3986 SP3987 SP3886 SP3887	https://twitter.com/TomEdwardsWild/status/1130151692966727680
Not Provided	17/05/2019	Willington	52.85219	-1.56078	SK22 SK32	SK2928 SK3028	https://twitter.com/Davidturner1967/status/1129448662122287105
16/05/2019	16/05/2019	Upton Warren	52.31076	-2.10391	SO96	SO9367 SO9366 SO9266 SO9267	https://twitter.com/Rich_Clifford/status/1129087948710920193
Not Provided	09/08/2018	Richmond Park, London	51.43638	-0.28226	TQ27 TQ17	TQ2072 TQ2171 TQ2173 TQ2071 TQ2073 TQ2074 TQ1970 TQ1972 TQ1973 TQ1974	https://twitter.com/bikebunny1/status/1027383346479947776
Not Provided	24/07/2018	River Ivel, Clifton, Bedfordshire	52.04115	-0.28064	TL13	TL1838 TL1839	https://twitter.com/AlasdairGordon1/status/1021755254671650817

Not Provided	22/06/2018	River Trent, Nottingham	52.94341	-1.12814	SK53 SK63	SK6139 SK6038 SK5938 SK5838 SK5737 SK5638 SK5637	https://twitter.com/lallawandavi/status/1021023284375085057
Not Provided	18/07/2018	Richmond Park, London	51.43638	-0.28226	TQ27 TQ17	TQ2072 TQ2171 TQ2173 TQ2071 TQ2073 TQ2074 TQ1970 TQ1972 TQ1973 TQ1974	https://twitter.com/issybryonyh/status/1019472449828802566
Not Provided	17/07/2018	Richmond Park, London	51.43638	-0.28226	TQ27 TQ17	TQ2072 TQ2171 TQ2173 TQ2071 TQ2073 TQ2074 TQ1970 TQ1972 TQ1973 TQ1974	https://twitter.com/issybryonyh/status/1019321145156407296
27/06/2018	27/06/2018	Pocklington Canal	53.89055	-0.86132	SE74	SE7042 SE7043 SE7144 SE7145 SE7245 SE7345 SE7444 SE7544 SE7644 SE7744 SE7844 SE7845 SE7946 SE7947	https://twitter.com/BagginsTim/status/1012077137178824704
Not Provided	27/06/2018	Eardington Nature Reserve	52.50578	-2.40886	SO79 SO78	SE7290 SO7289	https://twitter.com/PeterRockett/status/1012071980923867138
Not Provided	24/06/2018	Langford	52.05431	-0.27171	TL14	TL1840 TL1841	https://twitter.com/ravenfandango/status/1010957114947375104

21/06/2018	21/06/2018	Forest Farm, Cardiff	51.51959	-3.2488	ST18	ST1380	https://twitter.com/Billsyboi/status/1009814973047017477
Not Provided	20/06/2018	Langford	52.05431	-0.27171	TL14	TL1840 TL1841	https://twitter.com/ravenfandango/status/1009506393165062150
Not Provided	16/06/2018	Priory Park, Warwick	52.28502	-1.58482	SP26	SP2865	https://twitter.com/HalfasniceDave/status/1008049234824884224
Not Provided	11/06/2018	Ashleworth, River Severn	51.92372	-2.26397	SO82	SO8125 SO8124	https://twitter.com/JoPearce/status/1006071048746479617
Not Provided	21/05/2018	River Stour, Blandford Forum	50.85902	-2.17549	ST80	ST8806 ST8706 ST8707	https://twitter.com/richardbeechpix/status/998524750430425088
20/05/2018	20/05/2018	Canford Magna, Dorset	50.79048	-1.95346	SZ09	SZ0398 SZ0399	https://twitter.com/ballywhooo/status/998217714777886720
Not Provided	13/08/2017	Morden Hall Park	51.40319	-0.18676	TQ26	TQ2668	https://twitter.com/bikebunny1/status/896672303249051648
Not Provided	01/08/2017	Walthamstow Wetlands	51.58564	-0.04909	TQ38 TQ39	TQ3490 TQ3590 TQ3489 TQ3589 TQ3488 TQ3588 TQ3488	https://twitter.com/Phil3Production/status/892469992897339392
Not Provided	09/07/2017	Woodnewton	52.53587	-0.4775	TL09	TL0394	https://twitter.com/Haucksie/status/884083366169120768
Not Provided	30/06/2017	Brockholes nature reserve	53.77261	-2.63035	SD53	SD5830 SD5831	https://twitter.com/scotsgreyphoto/status/880891278845779968
Not Provided	20/06/2017	Hill holt wood	53.13484	-0.71251	SK86	SK8660	https://twitter.com/woodmanoliver1/status/877244569955561472
Not Provided	18/06/2017	Ferry Meadows	52.5661	-0.31053	TL19	TL1497 TL1597 TL1598 TL1498 TL1397 TL1398	https://twitter.com/papa_tilley/status/876459626388217856
Not Provided	11/06/2017	Syon Park	51.47848	-0.31387	TQ17	TQ1776 TQ1676	https://twitter.com/MsFrabby/status/873866396706504704
20/05/2017	20/05/2017	River Bure, Aylsham	52.80099	1.25781	TG12 TG22	TG1927 TG2027	https://twitter.com/Norf_Viz_Mig/status/869528323159994368

Not Provided	30/05/2017	Bewdley	52.37544	-2.31733	SO77	SO7875 SO7975 SO7974 SO7874 SO7774 SO7775	https://twitter.com/jasonkeyse/status/868804117912846336
Not Provided	30/05/2017	Bewdley	52.37544	-2.31733	SO77	SO7875 SO7975 SO7974 SO7874 SO7774 SO7775	https://twitter.com/jasonkeyse/status/868803843282391040
Not Provided	13/05/2017	Harrold Odell Country Park	52.20261	-0.59565	SP95	SP9556 SP9656 SP9657 SP9557	https://twitter.com/Spraggy2009/status/863445883223986177
Not Provided	06/08/2016	Lytchett fields	50.73155	-2.0511	SY99	SY9692 SY9693	https://twitter.com/lewishambirder/status/761850440329953280
24/07/2016	24/07/2016	Frampton Pools	51.76555	-2.35763	SO70	SO7507	https://twitter.com/Nickgoatman/status/757290198627676160
16/07/2016	17/07/2016	Lunt Meadows	53.51943	-2.988	SD30	SD3402 SD3502 SD3403	https://twitter.com/robmlewis/status/754798292207362048
12/07/2016	12/07/2016	New Passage	51.57066	-2.65122	ST58	ST5585 ST5485	https://twitter.com/Bigjohnatyeo/status/752897921042055168
20/06/2016	20/06/2016	Jesus Collage Cambridge	52.2096	0.12421	TL45	TL4558	https://twitter.com/Spraggy2009/status/744912419546357760
19/06/2016	19/06/2016	Anglesey Abbey, Cambridgeshire	52.23704	0.2411	TL56	TL5362 TL5262 TL5261	https://twitter.com/CRCuthbert/status/744636522071269381
Not Provided	17/06/2016	Attenborough nature reserve	52.90397	-1.22525	SK53	SK5033 SK5133 SK5233 SK5134 SK5234 SK5235 SK5335	https://twitter.com/GroolyIrene/status/743830551967072256

Not Provided	14/06/2016	Walthamstow Wetlands	51.58564	-0.04909	TQ38 TQ39	TQ3490 TQ3590 TQ3489 TQ3589 TQ3488 TQ3588 TQ3488	https://twitter.com/006_neo/status/742737990481510400
Not Provided	13/06/2016	Walthamstow Wetlands	51.58564	-0.04909	TQ38 TQ39	TQ3490 TQ3590 TQ3489 TQ3589 TQ3488 TQ3588 TQ3488	https://twitter.com/006_neo/status/742414509441159168
Not Provided	04/06/2016	Moor Trust's Gadespring Cress Beds	51.74415	-0.48558	TL00	TL0505 TL0405 TL0506 TL0406 TL0305 TL0306 TL0304 TL0204 TL0205 TL0206	https://twitter.com/DanFWildlife/status/738895614566670336
Not Provided	28/05/2016	Lackford Lake, Suffolk Wildlife Trust	52.3062	0.64384	TL87	TL8070 TL8170 TL7970 TL7971 TL8071	https://twitter.com/AMeredithPhotos/status/736626091880816640
25/05/2016	25/05/2016	Isle of Wight Donkey Sanctuary	50.62759	-1.22491	SZ58	SZ5481 SZ5581	https://twitter.com/iwgreengym/status/735569407846518784
Not Provided	21/05/2016	Earsham, River Waveney Trust	52.44361	1.40595	TM38	TM3188	https://twitter.com/AndrewAtterwill/status/734084386275102720
Not Provided	13/05/2016	Walthamstow Wetlands	51.58564	-0.04909	TQ38 TQ39	TQ3490 TQ3590 TQ3489 TQ3589 TQ3488 TQ3588 TQ3488	https://twitter.com/006_neo/status/731141842574643200
Not Provided	04/08/2015	Castle Acre, King's Lynn	52.70375	0.69129	TF81	TF8115 TF8114 TF8215 TF8214	https://twitter.com/e1_clair/status/628662662776418305

21/07/2015	21/07/2015	Hartslock, Reading	51.50994	-1.11105	SU67	SU6179 SU6279	https://twitter.com/towner_tony/status/623568016916221952
Not Provided	10/06/2015	South Cerney	51.66863	-1.9124	SU09	SU0599 SU0498 SU0598 SU0698 SU0497 SU0597 SU0697 SU0496 SU0596 SU0696 SU0796 SU0595 SU0695 SU0795 SU0794	https://twitter.com/tpsafari/status/608619845492572160
Not Provided	07/06/2015	River Severn, Bewdley	52.37628	-2.31342	SO77	SO7875	https://twitter.com/jasonkeyse/status/607601167523299328
Not Provided	12/06/2014	Riverside Walk, Kingston upon Thames	51.41015	-0.30833	TQ16	TQ1769	https://twitter.com/ymtgardenwatch/status/477027866035912705
06/06/2014	06/06/2014	River Shep, Shepreth	52.11194	0.03379	TL34	TL3947 TL3948	https://twitter.com/ForShep/status/474950619904880641
Not Provided	31/05/2014	Frome Valley	51.5341	-2.48247	ST68	ST6681	https://twitter.com/tiger8lotus/status/472824589022928896
Not Provided	16/05/2014	Wanstead Park	51.56867	0.03787	TQ48	TQ4087 TQ4187 TQ4287	https://twitter.com/wildlife_id/status/467206510708551681
Not Provided	08/06/2013	Brockdale Nature Reserve	53.65077	-1.24289	SE51 SE41	SE4917 SE5017 SE5117 SE5116	https://twitter.com/ChrisMcLoughlin/status/343482503060926464
Not Provided	24/06/2018	Ilminster	50.92746	-2.91413	ST31	ST3614 ST3514 ST3414 ST3415 ST3515 ST3513 ST3613	https://twitter.com/mindful_walks/status/1010934424928649216

Not Provided	14/08/2012	Epping Forest	51.66519	0.05416	TL40 TQ49 TQ39	TL4300 TL4400 TQ4199 TQ4299 TQ4399 TQ4499 TQ4198 TQ4298 TQ4398 TQ4097 TQ4197 TQ4297 TQ4096 TQ4196 TQ3995 TQ4094	https://twitter.com/andrewknots/status/235278614210416640

Table A.2. Banded Demoiselle (Calopteryx splendens) occurrences manually extracted from Facebook.

Date Recorded	Date Posted	Location Description	Latitude	Longitude	10km Grid	1km Grid	Website
Not Provided	27/08/2021	Mottisfont	51.04001	-1.53651	SU32	SU3226 SU3227	https://www.facebook.com/photo/?fbid=5951451471592347&set=a.5951677708236390
Not Provided	15/12/2021	Fittleworth	50.96084	-0.56214	TQ01	TQ0119 TQ0118 TQ0018 TQ0019	https://www.facebook.com/photo/?fbid=118792707299413&set=pcb.118792747299409
Not Provided	15/12/2021	Fittleworth	50.96084	-0.56214	TQ01	TQ0119 TQ0118 TQ0018 TQ0019	https://www.facebook.com/photo/?fbid=118792710632746&set=pcb.118792747299409
Not Provided	24/06/2021	Lackford Lakes	52.3062	0.64384	TL87	TL8070 TL8170 TL7970 TL7971 TL8071	https://www.facebook.com/photo/?fbid=4137847266334989&set=pcb.4137847416334974
Not Provided	24/06/2021	Lackford Lakes	52.3062	0.64384	TL87	TL8070 TL8170 TL7970 TL7971 TL8071	https://www.facebook.com/photo/?fbid=4137847273001655&set=pcb.4137847416334974
Not Provided	02/08/2021	Felmersham Gravel Pits	52.21505	-0.55115	SP95	SP9958 SP9858	https://www.facebook.com/OliverAndrewsPhotography/photos/pb.100061598927235.-2207520000/4146857268695951/?type=3
Not Provided	03/08/3021	Felmersham Gravel Pits	52.21505	-0.55115	SP95	SP9958 SP9858	https://www.facebook.com/OliverAndrewsPhotography/photos/pb.100061598927235.-2207520000/4149761131738898/?type=3
Not Provided	27/07/2021	Southwick Wood	52.51391	-0.4849	TL09	TL0291 TL0391 TL0392	https://www.facebook.com/OliverAndrewsPhotography/photos/pb.100061598927235.-2207520000/4129693653745646/?type=3
18/07/2021	18/07/2021	Rodley Nature Reserve	53.82046	-1.6473	SE23	SE2336	https://www.facebook.com/photo/?fbid=10222605808628442&set=pcb.4172364902842530

10/05/2020	10/05/2020	Bishopstoke, River Itchen	50.96923	-1.33728	SU41	SU4619 SU4618	https://www.facebook.com/photo.php?fbid=2923668567750151&set=pb.100003211101898.-2207520000&type=3
Not Provided	11/06/2020	Blackpool Brook, Forest of Deab	51.77002	-2.50343	SO60	SO6408 SO6508 SO6507 SO6606	https://www.facebook.com/photo.php?fbid=2742678849302077&set=pb.100006799659889.-2207520000&type=3
Not Provided	17/06/2020	Cromford Canal	53.08418	-1.50458	SK35	SK3452 SK3353 SK3354 SK3355 SK3255 SK3155 SK3156 SK3056	https://www.facebook.com/photo/?fbid=10160209387018747&set=gm.1881752615293672
27/05/2020	27/05/2020	River Itchen	50.96923	-1.33728	SU41	SU4619 SU4618	https://www.facebook.com/photo.php?fbid=2967563956693945&set=pb.100003211101898.-2207520000&type=3
Not Provided	29/12/2020	Woolbeding Parkland	50.99612	-0.74757	SU82	SU8722 SU8822	https://www.facebook.com/photo/?fbid=2550727605228953&set=pcb.2550730961895284
20/05/2020	20/05/2020	Bishopstoke, River Itchen	50.96923	-1.33728	SU41	SU4619 SU4618	https://www.facebook.com/photo.php?fbid=2950356558414685&set=pb.100003211101898.-2207520000&type=3
Not Provided	18/06/2020	Wharf Stream Walk, Eynsham	51.77639	-1.35775	SP40	SP4408	https://www.facebook.com/TheEynshamSnapper/photos/pb.100063573100295.-2207520000/972694823151345/?type=3
Not Provided	02/09/2020	River Crouch, Wickford	51.61458	0.52395	TQ79	TQ7393 TQ7493 TQ7593 TQ7594 TQ7694	https://www.facebook.com/photo/?fbid=3473943289293340&set=p.3473943289293340
Not Provided	02/09/2020	River Crouch, Wickford	51.61458	0.52395	TQ79	TQ7393 TQ7493 TQ7593 TQ7594 TQ7694	https://www.facebook.com/photo/?fbid=3473938429293826&set=p.3473938429293826

Not Provided	14/08/2020	Gavray Meadows	51.89566	-1.13332	SP52	SP5922	https://www.facebook.com/photo/?fbid=3468046479893886&set=pcb.3468048379893696
Not Provided	05/08/2020	Southwick Wood	52.51391	-0.4849	TL09	TL0291 TL0391 TL0392	https://www.facebook.com/OliverAndrewsPhotography/photos/pb.100061598927235.-2207520000/3130105040371184/?type=3
Not Provided	31/07/2020	Wilberfoss	53.94902	-0.88874	SE75	SE7351 SE7251 SE7250 SE7350	https://www.facebook.com/photo/?fbid=10158148648591187&set=pcb.276729166953644
Not Provided	25/07/2020	Middleton Lakes	52.59403	-1.70086	SP19 SP29 SK20	SP1998 SP2098 SP2099 SK2000	https://www.facebook.com/andrewmwildlife/photos/pb.100071151913546.-2207520000/101368595007826/?type=3
Jun-20	20/07/2020	Titchmarsh Nature Reserve	52.41276	-0.52114	TL08	TL0080 TL0081 TL0180	https://www.facebook.com/OliverAndrewsPhotography/photos/pb.100061598927235.-2207520000/3084904634891225/?type=3
Not Provided	14/07/2020	Fountain Inn, Ashurst	50.93252	-0.32153	TQ11	TQ1716 TQ1816	https://www.facebook.com/photo.php?fbid=4632978590052938&set=pb.100000225273346.-2207520000&type=3
01/07/2020	01/07/2020	Cuerden	53.70895	-2.66558	SD52	SD5624 SD5623 SD5524 SD5523	https://www.facebook.com/photo/?fbid=735408513883499&set=p.735408513883499

Not Provided	01/07/2020	Knepp	50.9835	-0.347	TQ12	TQ1523 TQ1522 TQ1521 TQ1520 TQ1320 TQ1321 TQ1420 TQ1421 TQ1623 TQ1622 TQ1621 TQ1620	https://www.facebook.com/photo.php?fbid=1924730427662369&set=pb.100003762910126.-2207520000&type=3
Not Provided	01/07/2020	Knepp Park	50.9835	-0.347	TQ12	TQ1523 TQ1522 TQ1521 TQ1520 TQ1320 TQ1321 TQ1420 TQ1421 TQ1623 TQ1622 TQ1621 TQ1620	https://www.facebook.com/photo/?fbid=1924718424330236&set=pcb.1924718854330193
Not Provided	03/08/2019	Melwood	53.43202	-2.89889	SJ49	SJ4093	https://www.facebook.com/meldrethhistory/photos/a.964156336997958/2390396511040593/?type=3
Not Provided	27/07/2019	Centenary Riverside Nature Reserve, Rotherham	53.42382	-1.36874	SK49	SK4292 SK4192	https://www.facebook.com/photo.php?fbid=2525732450825985&set=gm.2324242314363055&type=3
Not Provided	17/08/2019	Testwood Lakes	50.94017	-1.50545	SU31	SU3415 SU3515 SU3416	https://www.facebook.com/lefoto.peterlecointe/photos/a.738290286219214/2401384696576423/?type=3
Not Provided	25/09/2019	Middleton Lakes	52.59403	-1.70086	SP19 SP29 SK20	SP1998 SP2098 SP2099 SK2000	https://www.facebook.com/photo.php?fbid=2556308114415535&set=a.295731633806539&type=3

Not Provided	16/06/2019	Lackford Lakes	52.3062	0.64384	TL87	TL8070 TL8170 TL7970 TL7971 TL8071	https://www.facebook.com/photo.php?fbid=10156929455020932&set=a.10150309825465932&type=3
Not Provided	11/07/2019	Barmpton Lane	54.54721	-1.51505	NZ31	NZ3117 NZ3116	https://www.facebook.com/photo.php?fbid=1791390100964285&set=a.199346073502037&type=3
Not Provided	20/05/2019	Radipol	50.63002	-2.46882	SY68	SY6781 SY6681 SY6682 SY6782	https://www.facebook.com/photo.php?fbid=2240136339388695&set=a.738565382879139&type=3
Not Provided	24/07/2019	North Cave Wetlands	53.78532	-0.66315	SE83	SE8832 SE8833 SE8732 SE8733	https://www.facebook.com/photo.php?fbid=321092968795853&set=a.321094388795711&type=3
21/07/2019	27/07/2019	Ripon (Ref SE 31522 72280)	54.14539	-1.51827	SE37	SE3172	https://www.facebook.com/photo.php?fbid=10214612816181659&set=a.10214612798061206&type=3
15/07/2019	25/09/2019	Middleton Lakes	52.59403	-1.70086	SP19 SP29 SK20	SP1998 SP2098 SP2099 SK2000	https://www.facebook.com/photo.php?fbid=2556229887756691&set=a.295731633806539&type=3
Not Provided	28/09/2019	RSPB Sandy	52.1175	-0.26309	TL14	TL1847 TL1947 TL1848 TL1948	https://www.facebook.com/scperkinsphotography/photos/a.123613532368636/123744225688900/?type=3
28/05/2019	07/07/2019	Aldford, River Dee	53.12975	-2.87594	SJ45	SJ4159	https://www.facebook.com/photo.php?fbid=2364398120247239&set=gm.2303367646543588&type=3
27/07/2019	27/07/2019	Ref SE 31393 72919	54.15104	-1.52148	SE37	SE3172	https://www.facebook.com/photo.php?fbid=10214612861062781&set=a.10214612798061206&type=3
Not Provided	30/05/2019	Rooksbury	51.19865	-1.49253	SU34	SU3544	https://www.facebook.com/photo.php?fbid=10156759234558557&set=gm.2298909083509268&type=3

06/05/2019	06/05/2019	Whaddon	52.09698	-0.03208	TL34	TL3446 TL3546	https://www.facebook.com/photo.php?fbid=2649465271791430&set=gm.1439945839481865&type=3
Not Provided	23/07/2019	Tipton St John	50.71863	-3.28682	SY09	SY0991 SY0891 SY0892 SY0992	https://www.facebook.com/photo.php?fbid=10220553949665152&set=a.10202538314085522&type=3
08/07/2019	13/07/2019	Sadbury	52.0392	0.73296	TL84	TL8642 TL8742 TL8842 TL8641 TL8741 TL8841 TL8640 TL8740 TL8840 TL8940 TL8739 TL8839 TL8939	https://www.facebook.com/photo.php?fbid=10218596082349434&set=a.10208407224034344&type=3
Not Provided	02/08/2019	Isleworth	51.46937	-0.32775	TQ17	TQ1576 TQ1676 TQ1575 TQ1675 TQ1575 TQ1675	https://www.facebook.com/photo.php?fbid=10216543758124024&set=gm.2238716846226336&type=3
Not Provided	17/07/2019	Swillbrook	53.80661	-2.78419	SD43	SD4834	https://www.facebook.com/photo.php?fbid=10217861615319390&set=gm.461471461067674&type=3
29/07/2019	01/08/2019	Middleton Lakes	52.59403	-1.70086	SP19 SP29 SK20	SP1998 SP2098 SP2099 SK2000	https://www.facebook.com/photo.php?fbid=1916546611779386&set=gm.1572415392894064&type=3
Not Provided	06/07/2019	Low Barns Nature Reserve	54.67605	-1.75079	NZ13	NZ1631 NZ1531	https://www.facebook.com/photo.php?fbid=2348933815347446&set=gm.2157352831231038&type=3
Not Provided	23/05/2019	Titchmarsh Nature Reserve	52.41276	-0.52114	TL08	TL0080 TL0081 TL0180	https://www.facebook.com/photo.php?fbid=2755732641113088&set=gm.2348744532075690&type=3

23/07/2019	23/07/2019	Wicken Fen	52.30423	0.28421	TL57 TL56	TL5670 TL5570 TL5470 TL5769 TL5669 TL5569 TL5469 TL5668 TL5568 TL5667	https://www.facebook.com/photo.php?fbid=2363766090366539&set=gm.2128404833937625&type=3
Not Provided	18/07/2019	Winnall Moors	51.07625	-1.30139	SU42 SU43	SU4829 SU4830 SU4930 SU4831 SU4932	https://www.facebook.com/photo.php?fbid=2235352836577242&set=gm.822860304774363&type=3
Not Provided	22/08/2019	Batford Springs Nature Reserve	51.82248	-0.33788	TL11	TL1415 TL1414	https://www.facebook.com/photo.php?fbid=10156465059398461&set=gm.2918189851556521&type=3
Not Provided	28/05/2019	Sale Water Park	53.43506	-2.30112	SJ89	SJ8092 SJ7992 SJ8093 SJ7993	https://www.facebook.com/photo.php?fbid=10113815935082265&set=gm.2272264943012874&type=3
Not Provided	27/07/2019	Hall Place	51.4463	0.16188	TQ57	TQ5074	https://www.facebook.com/photo.php?fbid=2437430076311095&set=gm.2399033450420749&type=3
Not Provided	26/06/2019	River Aire between Rodley and Apperley Bridge	53.83942	-1.68102	SE23 SE13	SE2236 SE2137 SE2138 SE2038 SE1937	https://www.facebook.com/photo.php?fbid=2402987129767828&set=gm.2287140414698331&type=3
17/05/2019	17/05/2019	Tees at Yarm	54.5122	-1.35644	NZ41	NZ4112 NZ4212 NZ4113	https://www.facebook.com/photo.php?fbid=10157226359489819&set=gm.1512096425592628&type=3
Not Provided	05/07/2019	River Wye, Lydbrook	51.8496	-2.58333	SO61 SO51	SO6017 SO6016 SO5916 SO5917	https://www.facebook.com/photo.php?fbid=2415503482019617&set=gm.2302450299847480&type=3
Not Provided	05/06/2019	Snakeholme Pit	53.23009	-0.33022	TF17	TF1171	https://www.facebook.com/photo.php?fbid=2739338216108049&set=gm.10725201662707

							5&type=3
08/07/2019	08/07/2019	Rooksbury	51.19865	-1.49253	SU34	SU3544	https://www.facebook.com/photo.php?fbid=10156859354368557&set=gm.2366725266727649&type=3
05/08/2019	05/08/2019	White Coppice	53.66665	-2.5842	SD61	SD6119 SD6118	https://www.facebook.com/photo.php?fbid=10215923856714194&set=gm.902293256792184&type=3
26/05/2019	26/05/2019	West Amesbury, River Avon	51.1712	-1.79752	SU14	SU1441	https://www.facebook.com/photo.php?fbid=10214547869074908&set=gm.1282907278535258&type=3
23/06/2019	24/06/2019	Ufford	52.12212	1.35346	TM25 TM35	TM2952 TM3052 TM2953 TM3053	https://www.facebook.com/photo.php?fbid=10157223339883150&set=gm.1477722332370882&type=3
12/05/2019	12/05/2019	National Memorial Arboretum	52.73109	-1.7248	SK11	SK1814 SK1815 SK1915	https://www.facebook.com/photo.php?fbid=1092248294291731&set=a.289402144576354&type=3
19/06/2019	19/06/2019	Wheldrake Ings	53.88654	-0.93228	SE64	SE6944 SE7044 SE6943 SE7043	https://www.facebook.com/photo.php?fbid=10162239313530529&set=gm.2275152565897116&type=3
Not Provided	16/11/2010	National Botanic Garden of Wales	51.84254	-4.14635	SN51	SN5117 SN5217 SN5218 SN5118	https://www.facebook.com/photo.php?fbid=10150127108159128&set=a.10150127107344128&type=3
Not Provided	26/05/2010	River Witham, Bassingham	53.12942	-0.64307	SK95	SK9059 SK9060	https://www.facebook.com/photo.php?fbid=106967946016296&set=a.101837683195989&type=3

Not Provided	25/06/2010	Richmond Park, London	51.43638	-0.28226	TQ27 TQ17	TQ2072 TQ2171 TQ2173 TQ2071 TQ2073 TQ2074 TQ1970 TQ1972 TQ1973 TQ1974	https://www.facebook.com/photo.php?fbid=1287451681276&set=a.1170878247013&type=3
Not Provided	10/07/2010	Lullingstone Castle Gardens	51.35882	0.19631	TQ56	TQ5264 TQ5364	https://www.facebook.com/photo.php?fbid=1492878449354&set=a.1492730885665&type=3
Not Provided	25/06/2010	Beverley Brook, Richmond Park	51.44683	-0.25459	TQ27	TQ2173 TQ2172	https://www.facebook.com/photo.php?fbid=1287451641275&set=a.1170878247013&type=3
Not Provided	28/06/2010	River Sow, Tixall	52.79801	-2.03074	SJ92	SJ9722 SJ9822 SJ9721 SJ9821	https://www.facebook.com/photo.php?fbid=134621513223028&set=a.104080876277092&type=3
Not Provided	07/07/2010	River Blackwater near Totton	50.92746	-1.47894	SU31	SU3712 SU3713 SU3613 SU3614 SU3515 SU3415	https://www.facebook.com/photo.php?fbid=458399147288&set=a.447029647288&type=3
Not Provided	06/08/2010	Wood Lane Nature Reserve	52.89164	-2.85639	SJ43	SJ4232 SJ4233	https://www.facebook.com/ShropshireWildlife/photos/a.111509825560128/129467443764366/?type=3
Not Provided	21/05/2010	Thrapston Walk	52.39573	-0.5407	SP97	SP9978	https://www.facebook.com/photo.php?fbid=405056785002&set=a.274743975002&type=3
Not Provided	21/05/2010	Thrapston Walk	52.39573	-0.5407	SP97	SP9978	https://www.facebook.com/photo.php?fbid=405056770002&set=a.274743975002&type=3
25/05/2011	25/05/2011	Winnall Moors	51.07625	-1.30139	SU42 SU43	SU4829 SU4830 SU4930 SU4831 SU4932	https://www.facebook.com/photo.php?fbid=10150252470735664&set=a.10150223123985664&type=3

Not Provided	19/05/2011	Parkbroom, Carlisle	54.92088	-2.8881	NY45	NY4358 NY4359	https://www.facebook.com/CarlisleNats/photos/a.406728006427/10150185241716428/?type=3
01/07/2010	07/04/2012	Balcombe Mills	50.91637	0.0324	TQ41	TQ4314 TQ4214 TQ4315 TQ4215	https://www.facebook.com/314554471933116/photos/a.316968788358351/326800624041834/?type=3
Not Provided	27/07/2012	River Salwarpe, Moors Pool, Upton Warren	52.30336	-2.09321	SO96	SO9367	https://www.facebook.com/photo.php?fbid=4430280800552&set=a.4328137407031&type=3
Not Provided	30/09/2012	Attenborough nature reserve	52.90397	-1.22525	SK53	SK5033 SK5133 SK5233 SK5134 SK5234 SK5235 SK5335	https://www.facebook.com/JackPerksPhotography/photos/a.197152673745287/227025660757988/?type=3
30/07/2012	30/07/2012	Brockdale Nature Reserve	53.65077	-1.24289	SE51 SE41	SE4917 SE5017 SE5117 SE5116	https://www.facebook.com/photo.php?fbid=10151115822247279&set=gm.163301313805486&type=3
21/08/2012	21/08/2012	Brockdale Nature Reserve	53.65077	-1.24289	SE51 SE41	SE4917 SE5017 SE5117 SE5116	https://www.facebook.com/photo.php?fbid=10151163887492279&set=gm.170659259736358&type=3
Not Provided	30/07/2012	Brockdale Nature Reserve	53.65077	-1.24289	SE51 SE41	SE4917 SE5017 SE5117 SE5116	https://www.facebook.com/photo.php?fbid=10151115834062279&set=gm.163302700472014&type=3
24/07/2012	25/07/2012	Stodmarsh National Nature Reserve (Stodge Marsh)	51.31265	1.18915	TR26	TR2161 TR2261 TR2262 TR2362	https://www.facebook.com/RWFordPhotography/photos/a.375946675805936/391896950877575/?type=3
Not Provided	01/12/2012	Machynlleth	52.591	-3.85045	SH70	SH7400 SH7500 SH7600 SH7401 SH7501	https://www.facebook.com/photo.php?fbid=10151538820081258&set=a.10150814347491258&type=3

Not Provided	05/07/2012	River Alt, Lunt, Merseyside	53.51759	-2.97861	SD30	SD3403 SD3502	https://www.facebook.com/lancashirewildlifetrust/photos/a.136524139705944/460008467357508/?type=3
05/08/2012	30/05/2012	River Waveney near Bungay	52.45692	1.44071	TM38	TM3489 TM3389 TM3390	https://www.facebook.com/photo.php?fbid=3103917977412&set=a.2858594524479&type=3
Not Provided	21/05/2012	Cuerden Valley Park (near River)	53.70705	-2.65737	SD52	SD5624 SD5623 SD5622	https://www.facebook.com/photo.php?fbid=464613560221342&set=a.464612150221483&type=3
Not Provided	24/07/2012	Woolbeding Parkland	50.99612	-0.74757	SU82	SU8722 SU8822	https://www.facebook.com/photo.php?fbid=387145104672642&set=a.177815675605587&type=3
Not Provided	09/03/2012	Thurlby Fen	52.73319	-0.34254	TF11	TF1216 TF1116 TF1115	https://www.facebook.com/photo.php?fbid=10150649388281738&set=o.171779319576810&type=3
Not Provided	05/07/2012	Low Hall Nature Reserve, near Hindley	53.52452	-2.58741	SD60	SD6003 SD6103	https://www.facebook.com/ihlhq/photos/a.350682271672143/359359070804463/?type=3
Not Provided	04/06/2021	Geldeston	52.47274	1.52028	TM39	TM3991 TM3992 TM3891 TM3892	https://www.facebook.com/photo.php?fbid=10223959618351345&set=gm.2202461379889459&type=3
Not Provided	18/06/2021	Geldeston	52.47274	1.52028	TM39	TM3991 TM3992 TM3891 TM3892	https://www.facebook.com/photo.php?fbid=10224053608701045&set=gm.2756590801153524&type=3
Not Provided	02/07/2021	Woods Mill	50.90852	-0.26732	TQ21	TQ2113 TQ2213	https://www.facebook.com/photo.php?fbid=2884384978489542&set=gm.2924047357850693&type=3

Not Provided	28/07/2021	Hilton Derbyshire	52.87311	-1.63609	SK23 SK22	SK2429 SK2529 SK2629 SK2430 SK2530 SK2630 SK2431 SK2531 SK2631 SK2432 SK2532	https://www.facebook.com/photo.php?fbid=116463440715252&set=gm.921400618590043&type=3
Not Provided	06/06/2021	Chester Le Street, Riverside	54.85791	-1.56538	NZ25	NZ2852 NZ2851 NZ2850 NZ2849 NZ2750 NZ2749 NZ2751	https://www.facebook.com/photo.php?fbid=10225526726377873&set=a.10202283153703083&type=3
Not Provided	15/07/2021	Longham Lakes	50.78078	-1.91473	SZ09	SZ0698 SZ0697 SZ0597 SZ0598	https://www.facebook.com/photo.php?fbid=4209939082400290&set=a.830597000334532&type=3
15/08/2021	15/08/2021	Sadbury	52.0392	0.73296	TL84	TL8642 TL8742 TL8842 TL8641 TL8741 TL8841 TL8640 TL8740 TL8840 TL8940 TL8739 TL8839 TL8939	https://www.facebook.com/photo.php?fbid=10225929150591557&set=a.10204734452017339&type=3
Not Provided	16/06/2021	Whiteadder near Chirnside	55.79294	-2.21617	NT85	NT8556 NT8555 NT8455 NT8655 NT8654 NT8754 NT8755	https://www.facebook.com/photo.php?fbid=4303482759696149&set=gm.4622912791070121&type=3

Not Provided	10/06/2021	Geldeston	52.47274	1.52028	TM39	TM3991 TM3992 TM3891 TM3892	https://www.facebook.com/photo.php?fbid=10223998933094189&set=gm.2748338361978768&type=3
Not Provided	02/07/2021	Maesycrugiau	52.04819	-4.22512	SN44	SN4740 SN4741 SN4840 SN4841	https://www.facebook.com/photo.php?fbid=4097607477025469&set=gm.1672743639601845&type=3
Not Provided	09/06/2021	Geldeston	52.47274	1.52028	TM39	TM3991 TM3992 TM3891 TM3892	https://www.facebook.com/photo.php?fbid=10223994554264721&set=gm.2206395832829347&type=3
Not Provided	06/09/2021	Reigate Priory	51.23444	-0.20602	TQ25	TQ2550 TQ2549 TQ2449	https://www.facebook.com/photo.php?fbid=3037346003210427&set=a.1403653873246323&type=3
Not Provided	20/07/2021	Longham	50.78627	-1.90661	SZ09	SZ0698 SZ0697 SZ0797 SZ0798 SZ0699 SZ0799	https://www.facebook.com/photo.php?fbid=4224909797569885&set=a.830597000334532&type=3
Not Provided	08/06/2021	Reddish Vale	53.4373	-2.14553	SJ99	SJ9093 SJ9193	https://www.facebook.com/photo.php?fbid=10223417593660357&set=a.10208688930452982&type=3
22/07/2021	23/07/2021	Sadbury	52.0392	0.73296	TL84	TL8642 TL8742 TL8842 TL8641 TL8741 TL8841 TL8640 TL8740 TL8840 TL8940 TL8739 TL8839 TL8939	https://www.facebook.com/photo.php?fbid=10225767745796538&set=a.10204734452017339&type=3
28/05/2021	29/05/2021	RSPB Strumpshaw Fen	52.60337	1.45373	TG30	TG3405 TG3406 TG3305 TG3306 TG3307	https://www.facebook.com/photo.php?fbid=4169009476453715&set=gm.3806178019493623&type=3

Not Provided	28/06/2021	River Irwell, Salford	53.47482	-2.28595	SJ89 SD80 SJ79	SJ8299 SJ8298 SJ8399 SD8200 SD8100 SJ7898 SJ7998 SJ7997	https://www.facebook.com/photo.php?fbid=954492658673649&set=gm.1504455976568691&type=3
Not Provided	20/06/2021	Geldeston	52.47274	1.52028	TM39	TM3991 TM3992 TM3891 TM3892	https://www.facebook.com/photo.php?fbid=10224065811326103&set=gm.2215169161952014&type=3
Not Provided	10/06/2021	Earlswood Common	51.22367	-0.17777	TQ24	TQ2648 TQ2748 TQ2749	https://www.facebook.com/photo.php?fbid=2971374176474277&set=a.1403653873246323&type=3
01/06/2021	02/06/2021	Lakenheath Fen	52.44582	0.50522	TL78	TL7286 TL7186 TL7086 TL7085 TL6986 TL6985 TL6886 TL6885 TL6785	https://www.facebook.com/photo.php?fbid=4179211612100168&set=gm.3815924171852341&type=3
Not Provided	14/06/2021	Middleton Lakes	52.59403	-1.70086	SP19 SP29 SK20	SP1998 SP2098 SP2099 SK2000	https://www.facebook.com/photo.php?fbid=3522276324539732&set=gm.2118727584937017&type=3
Not Provided	17/07/2021	Kirkcudbright	54.83798	-4.04862	NX65	NX6750 NX6850 NX6751 NX6851 NX6950 NX6951 NX6752 NX6852 NX6952	https://www.facebook.com/photo.php?fbid=4321955587871554&set=gm.10158807590032950&type=3
02/07/2021	02/07/2021	Cuerden Valley Park	53.70895	-2.66558	SD52	SD5624 SD5623 SD5524 SD5523	https://www.facebook.com/chorleynats/photos/a.1269507299803117/4101143416639477/?type=3
Not Provided	20/05/2021	Hall Place	51.4463	0.16188	TQ57	TQ5074	https://www.facebook.com/photo.php?fbid=1

							0226239583520101&set=gm.1367940990246878&type=3
10/06/2021	10/06/2021	Kennet and Avon Canal near Kintbury	51.40174	-1.45083	SU36	SU3767 SU3867 SU3967	https://www.facebook.com/photo.php?fbid=10226405684395556&set=gm.584635046252813&type=3
Not Provided	12/06/2021	River Ure near Ripon	54.14256	-1.5127	SE37	SE3172 SE3271 SE3270	https://www.facebook.com/photo.php?fbid=10158506416249615&set=gm.4074923082586713&type=3
Not Provided	10/06/2021	Geldeston	52.47274	1.52028	TM39	TM3991 TM3992 TM3891 TM3892	https://www.facebook.com/photo.php?fbid=10223998925614002&set=gm.2748337721978832&type=3
04/08/2021	04/08/2021	Middleton Lakes	52.59403	-1.70086	SP19 SP29 SK20	SP1998 SP2098 SP2099 SK2000	https://www.facebook.com/photo.php?fbid=2614037652225410&set=a.1481826952113158&type=3
Not Provided	16/07/2021	Meads Eco Park, Bridgewater	51.1233	-3.00984	ST23	ST2936	https://www.facebook.com/photo.php?fbid=986991922061609&set=a.114745905952886&type=3
Not Provided	11/06/2018	Abney Hall Park	53.39853	-2.21238	SJ88	SJ8689 SJ8688 SJ8588 SJ8589	https://www.facebook.com/555455051239144/photos/a.555642934553689/1656732327778072/?type=3

24/06/2018	26/06/2018	Minsmere	52.25037	1.60639	TM46	TM4767 TM4766 TM4765 TM4665 TM4666 TM4566 TM4667 TM4567 TM4467 TM4469 TM4569 TM4669 TM4769 TM4468 TM4568 TM4668 TM4768	https://www.facebook.com/ilovebirdsandnature/photos/a.1587079354856811/2192484130982994/?type=3
Not Provided	28/06/2018	Pulborough Brooks	50.93878	-0.49251	TQ01	TQ0616 TQ0516 TQ0416 TQ0517 TQ0617	https://www.facebook.com/photo.php?fbid=2536969189650273&set=a.101109186569631&type=3
Not Provided	29/07/2018	Middleton Lakes	52.59403	-1.70086	SP19 SP29 SK20	SP1998 SP2098 SP2099 SK2000	https://www.facebook.com/photo.php?fbid=1420099221424130&set=gm.1297646433704296&type=3
Not Provided	02/06/2018	River Penk, Lower Drayton Farm	52.73879	-2.10048	SJ91	SJ9315	https://www.facebook.com/photo.php?fbid=1525751404203721&set=a.147416135370595&type=3
Not Provided	22/08/2018	Oxnead	52.76727	1.30605	TG22	TG2224 TG2324 TG2223 TG2323	https://www.facebook.com/photo.php?fbid=10156328909268382&set=gm.2146718408880157&type=3

24/05/2018	25/05/2018	Chainbridge Nature Reserve, Lound (Idle Valley Nature Reserve)	53.36241	-0.93915	SK68 SK78	SK6986 SK6985 SK6984 SK6983 SK6884 SK6883 SK7087 SK7086 SK7085 SK7084 SK7187 SK7186 SK7185	https://www.facebook.com/photo.php?fbid=389738028195332&set=gm.1181000448709740&type=3
Not Provided	01/07/2018	Highwoods	51.9095	0.90891	TM02	TM0028 TM0027 TM0026 TM0126 TM0127	https://www.facebook.com/photo.php?fbid=10155954365134735&set=gm.1253891278078921&type=3
Not Provided	19/09/2018	Conigre Mead, Melksham	51.37296	-2.14439	ST96	ST9063	https://www.facebook.com/photo.php?fbid=10157862181486038&set=gm.1118785974947390&type=3
09/07/2018	09/07/2018	River Maden	51.45045	-2.0585	ST97	ST9373 ST9473 ST9573 ST9572 ST9672 ST9772 ST9771 ST9871 ST9870 ST9970	https://www.facebook.com/SoniaHillphotography/photos/a.208674033044433/268086400436529/?type=3
07/06/2018	07/06/2018	Burnley Canal, near bus station	53.78761	-2.23799	SD83	SD8432	https://www.facebook.com/photo.php?fbid=2269496039733217&set=gm.2262099747350846&type=3

Not Provided	08/07/2018	Tewkesbury	51.99273	-2.15547	SO83 SO93	SO8834 SO8833 SO8832 SO8831 SO8934 SO8933 SO8932 SO8931 SO9034 SO9033 SO9032 SO9031	https://www.facebook.com/photo.php?fbid=1745446788870976&set=a.120606974688307&type=3
04/07/2018	04/07/2018	Red Lodge	52.30693	0.4944	TL67 TL77 TL66 TL76	TL6970 TL7070 TL6969 TL7069	https://www.facebook.com/photo.php?fbid=10156707551601414&set=gm.1045742248918430&type=3
01/07/2018	05/07/2018	Middleton Lakes	52.59403	-1.70086	SP19 SP29 SK20	SP1998 SP2098 SP2099 SK2000	https://www.facebook.com/photo.php?fbid=1385987914835261&set=gm.1271882392947367&type=3
Not Provided	07/06/2018	Lower Drayton Farm, Penkridge	52.73879	-2.10499	SJ91	SJ9315 SJ9215	https://www.facebook.com/photo.php?fbid=1530552767056918&set=gm.986422081516106&type=3
Not Provided	04/05/2018	Footscray Meadows	51.42458	0.12528	TQ47	TQ4771 TQ4871 TQ4772 TQ4872	https://www.facebook.com/photo.php?fbid=10216637170465776&set=gm.566173620423623&type=3
09/07/2018	09/07/2018	Castlefields, Calne	51.43616	-2.00656	ST97	ST9971 ST9970	https://www.facebook.com/photo.php?fbid=10156655506157743&set=gm.1051004608392194&type=3
Not Provided	13/06/2018	Winnall Moors	51.07625	-1.30139	SU42 SU43	SU4829 SU4830 SU4930 SU4831 SU4932	https://www.facebook.com/photo.php?fbid=10155386065168414&set=gm.568250513568678&type=3
27/05/2018	27/05/2018	River Stour at Throop	50.7587	-1.82871	SZ19	SZ1195 SZ1295	https://www.facebook.com/photo.php?fbid=2518468691512636&set=gm.1238440442958229&type=3

10/06/2018	10/06/2018	River Wye near Hereford	52.05242	-2.71595	SO43 SO53	SO4939 SO5039 SO5139 SO5138	https://www.facebook.com/photo.php?fbid=1137310179742225&set=gm.1246961835439423&type=3
25/06/2018	26/06/2018	Greylake Nature Reserve	51.11066	-2.86609	ST33	ST3934 ST3935	https://www.facebook.com/photo.php?fbid=1732337556842952&set=gm.2193132764254584&type=3
02/07/2018	02/07/2018	Topsham	50.68381	-3.46507	SX98	SX9589 SX9588 SX9587 SX9689 SX9688 SX9687 SX9789 SX9788 SX9787	https://www.facebook.com/photo.php?fbid=2564370520455541&set=gm.2086556024948112&type=3
Not Provided	23/05/2018	Fairburn Ings	53.74072	-1.30713	SE42	SE4727 SE4627 SE4527 SE4427 SE4426 SE4327 SE4326	https://www.facebook.com/photo.php?fbid=10210337922357642&set=p.10210337922357642&type=3
07/06/2018	08/06/2018	Middleton Lakes	52.59403	-1.70086	SP19 SP29 SK20	SP1998 SP2098 SP2099 SK2000	https://www.facebook.com/photo.php?fbid=1834125046633849&set=a.295731633806539&type=3
29/05/2018	29/05/2018	River Tame at Lea Marston Lakes	52.553	-1.68663	SP29	SP2194 SP2195	https://www.facebook.com/photo.php?fbid=1823721111007576&set=a.295731633806539&type=3
May/June	2708/2018	Shugborough Estate	52.79095	-2.0152	SJ92	SJ9921 SJ9821	https://www.facebook.com/photo.php?fbid=2116335145051645&set=a.104080876277092&type=3
13/06/2017	13/06/2017	Woodwalton Fen	52.44524	-0.19163	TL28	TL2384 TL2385 TL2285 TL2284 TL2283 TL2383	https://www.facebook.com/woodwaltonandholmefens/photos/a.1678386452404852/1944574315786063/?type=3
Not Provided	05/07/2017	Wolseley Centre	52.7794	-1.96544	SK02	SK0220	https://www.facebook.com/StaffsWildlife/pho

							tos/a.10150172005566796/10155457703721796/?type=3
Not Provided	17/06/2017	Coppermill Stream, Walthamstow Wetlands	51.58108	-0.0541	TQ38	TQ3489 TQ3589 TQ3488 TQ3588 TQ3478	https://www.facebook.com/WalthamstowWetlands/photos/a.838111472915834/1474188072641501/?type=3
08/07/2017	09/07/2017	RHS Garden Wisley	51.31248	-0.47353	TQ05	TQ0658 TQ0657 TQ0659	https://www.facebook.com/photo.php?fbid=10154803411493435&set=p.10154803411493435&type=3
16/05/2017	16/05/2017	Attenborough nature reserve	52.90397	-1.22525	SK53	SK5033 SK5133 SK5233 SK5134 SK5234 SK5235 SK5335	https://www.facebook.com/147891378588086/photos/a.307841322593090/1449905455053332/?type=3
Not Provided	09/08/2017	Canterbury	51.27966	1.08072	TR15	TR1356 TR1456 TR1556 TR1656 TR1357 TR1457 TR1557 TR1657 TR1358 TR1458 TR1558 TR1658 TR1359 TR1459 TR1559 TR1659	https://www.facebook.com/photo.php?fbid=1372891692780480&set=p.1372891692780480&type=3
Not Provided	01/07/2017	Spencer Road Wetlands	51.38372	-0.16307	TQ26	TQ2766	https://www.facebook.com/photo.php?fbid=10155537925174703&set=a.10155235811474703&type=3
Not Provided	22/06/2017	Doxey Marshes	52.81669	-2.13608	SJ92 SJ89	SJ9025 SJ9024 SJ9124 SJ9123 SJ8924	https://www.facebook.com/photo.php?fbid=10155336802668150&set=p.10155336802668150&type=3

Not Provided	08/07/2017	Carlton Marshes, Suffolk	52.4799	1.68093	TM59	TM5091 TM4991 TM4992 TM5092 TM5192 TM4993 TM5093 TM4994 TM5094	https://www.facebook.com/insidesuffolk/photos/a.1586120128348126/1748038435489627/?type=3
Not Provided	23/06/2017	Dalbeattie	54.93217	-3.82086	NX86	NX8262 NX8362 NX8462 NX8261 NX8361 NX8461 NX8260 NX8360 NX8460 NX8259 NX8359 NX8459	https://www.facebook.com/BritishDragonflySoc/photos/a.480292065365219/1473964105998005/?type=3
12/07/2017	12/07/2017	Willington	52.85315	-1.56534	SK22	SK2829 SK2929 SK3029 SK2828 SK2928 SK3028	https://www.facebook.com/photo.php?fbid=1601175159927146&set=p.1601175159927146&type=3
09/05/2017	10/05/2017	Troublefield, Dorset	50.77745	-1.82294	SZ19	SZ1297	https://www.facebook.com/hampshiredragonflies/photos/a.1493532690972992/1846764925649765/?type=3
Not Provided	13/07/2017	Kirkby Gravel Pits, Lincolnshire Wildlife Trust	53.13369	-0.1501	TF26	TF2361 TF2360	https://www.facebook.com/photo.php?fbid=10209447559972837&set=a.10208235727077772&type=3
15/06/2017	15/06/2017	Hooks Hall Pond	51.55297	0.1829	TQ58	TQ5186	https://www.facebook.com/photo.php?fbid=676942109157473&set=a.260598230791865&type=3
09/06/2017	10/06/2017	Lunt Meadows	53.51943	-2.988	SD30	SD3402 SD3502 SD3403	https://www.facebook.com/photo.php?fbid=131999030709733&set=gm.1015889876603044 2&type=3

31/08/2017	31/08/2017	Middleton Lakes	52.59403	-1.70086	SP19 SP29 SK20	SP1998 SP2098 SP2099 SK2000	https://www.facebook.com/photo.php?fbid=1546614728718217&set=a.295731633806539&type=3
Not Provided	27/06/2017	Thames at Goring	51.52233	-1.14192	SU68	SU5981 SU6081 SU5980 SU5979 SU6079	https://www.facebook.com/DominicMackenzieWildlifePhotography/photos/a.613549438777992/1131928283606769/?type=3
Jun-16	23/01/2017	Titchmarsh Nature Reserve	52.41276	-0.52114	TL08	TL0080 TL0081 TL0180	https://www.facebook.com/photo.php?fbid=403908896621018&set=a.116393762039201&type=3
Not Provided	25/09/2017	RHS Garden Wisley	51.31248	-0.47353	TQ05	TQ0658 TQ0657 TQ0659	https://www.facebook.com/photo.php?fbid=10155035864503435&set=gm.1037839396359180&type=3
28/05/2017	29/05/2017	Snakeholme Pit	53.23009	-0.33022	TF17	TF1171	https://www.facebook.com/photo.php?fbid=1869622079970000&set=gm.655707504618642&type=3
Not Provided	04/11/2017	Brockholes	53.77311	-2.63116	SD53	SD5830 SD5930 SD5931 SD5831 SD5731	https://www.facebook.com/photo.php?fbid=1871882292908298&set=gm.1328928090568250&type=3
13/08/2017	13/08/2017	Bradgers Hill, Luton	51.90312	-0.40887	TL02	TL0923 TL0924	https://www.facebook.com/bradgershill/photos/a.440799796086603/843032295863349/?type=3
Not Provided	28/08/2017	Sugar Mill Ponds, East York	53.68123	-0.94907	SE62	SE6921 SE6920	https://www.facebook.com/photo.php?fbid=10213629071538480&set=gm.1443308682414846&type=3

09/06/2017	09/06/2017	Leigh	53.49725	-2.5184	SD60 SJ69	SD6401 SD6501 SD6601 SD6701 SD6801 SD6400 SD6500 SD6600 SD6700 SD6800 SJ6499 SJ6599 SJ6699 SJ6799 SJ6899 SJ6498 SJ6598 SJ6698 SJ6798 SJ6898	https://www.facebook.com/photo.php?fbid=466237223713737&set=gm.458275151193999&type=3
Not Provided	22/09/2017	Brockdale Nature Reserve	53.65077	-1.24289	SE51 SE41	SE4917 SE5017 SE5117 SE5116	https://www.facebook.com/photo.php?fbid=844814862362436&set=gm.1293648647429528&type=3
Not Provided	30/05/2017	Throop	50.73731	-2.24695	SY89	SY8293 SY8292	https://www.facebook.com/photo.php?fbid=1974750149217829&set=gm.1025529160916026&type=3
11/08/2017	11/08/2017	Needham Lakes, Suffolk	52.15131	1.05901	TM05	TM0954	https://www.facebook.com/photo.php?fbid=10155506987413150&set=gm.1012862765523510&type=3
Not Provided	24/04/2017	Chiddingstone	51.18824	0.14263	TQ54	TQ5045 TQ4945	https://www.facebook.com/photo.php?fbid=821678787986571&set=gm.10155071641343046&type=3

Not Provided	03/06/2017	Wicken Fen	52.30423	0.28421	TL57 TL56	TL5670 TL5570 TL5470 TL5769 TL5669 TL5569 TL5469 TL5668 TL5568 TL5667	https://www.facebook.com/photo.php?fbid=10155279096868954&set=gm.963765170433270&type=3
30/05/2017	30/05/2017	Hassacarr Nature Reserve	53.95897	-0.97676	SE65	SE6751	https://www.facebook.com/photo.php?fbid=1825507457778376&set=gm.1354283381317377&type=3
Not Provided	07/07/2017	Carlton Marshes, Suffolk	52.4799	1.68093	TM59	TM5091 TM4991 TM4992 TM5092 TM5192 TM4993 TM5093 TM4994 TM5094	https://www.facebook.com/photo.php?fbid=10155690314082871&set=gm.1320033178142634&type=3
Not Provided	10/05/2017	Whaddon	52.09698	-0.03208	TL34	TL3446 TL3546	https://www.facebook.com/photo.php?fbid=1477803698957599&set=gm.1014092572059685&type=3
19/05/2017	20/05/2017	Hampton Loade, River Severn	52.47537	-2.37391	SO78	SO7486	https://www.facebook.com/photo.php?fbid=10209402590124880&set=gm.1180688902041749&type=3
Not Provided	21/06/2017	Southwick	50.83457	-0.23784	TQ20	TQ2306 TQ2406 TQ2506 TQ2305 TQ2405 TQ2505 TQ2304 TQ2404 TQ2504	https://www.facebook.com/photo.php?fbid=801972143314178&set=a.102628446581888&type=3
Not Provided	19/06/2017	Molessey Heath	51.39226	-0.37621	TQ16	TQ1267 TQ1367 TQ1266 TQ1366	https://www.facebook.com/photo.php?fbid=10155442263339224&set=a.10150254180184224&type=3

Not Provided	28/06/2017	Molessey Heath	51.39226	-0.37621	TQ16	TQ1267 TQ1367 TQ1266 TQ1366	https://www.facebook.com/BevilTSPhotos/photos/a.945258172250490/1226862400756731/?type=3
Not Provided	05/06/2016	Arley, Worcesteshire	52.41678	-2.34893	SO77	SO7579 SO7679 SO7580 SO7680	https://www.facebook.com/philipmoorephotography/photos/a.706349046089589/1144149282309561/?type=3
14/05/2016	14/05/2016	Throop	50.73731	-2.24695	SY89	SY8293 SY8292	https://www.facebook.com/photo.php?fbid=1434528633239986&set=gm.794170850718526&type=3
05/05/2016	09/06/2016	Titchmarsh Nature Reserve	52.41276	-0.52114	TL08	TL0080 TL0081 TL0180	https://www.facebook.com/photo.php?fbid=289319434746632&set=a.116393762039201&type=3
08/08/2016	08/08/2016	Otmoor	51.82108	-1.18647	SP51	SP5514 SP5614 SP5513 SP5613 SP5512 SP5612 SP5714 SP5712	https://www.facebook.com/photo.php?fbid=1249995921699593&set=a.195625303803332&type=3
07/07/2016	07/07/2016	Stodmarsh National Nature Reserve (Stodge Marsh)	51.31265	1.18915	TR26	TR2161 TR2261 TR2262 TR2362	https://www.facebook.com/photo.php?fbid=1017339821655130&set=gm.883769595083980&type=3
17/06/2016	17/06/2016	Stanwick	52.33098	-0.56638	SP97	SP9871 SP9771 SP9671 SP9670 SP9672 SP9772	https://www.facebook.com/photo.php?fbid=1130323636989603&set=a.101602099861767&type=3
10/08/2016	10/08/2016	Throop	50.73731	-2.24695	SY89	SY8293 SY8292	https://www.facebook.com/photo.php?fbid=1523375821021933&set=gm.843823715753239&type=3

05/07/2016	05/07/2016	Brandon Marsh	52.3771	-1.43086	SP37	SP3875 SP3975 SP3874 SP3974 SP3876 SP3976	https://www.facebook.com/photo.php?fbid=649116758577009&set=gm.823795534422724&type=3
22/06/2016	22/06/2016	Crickdale North Meadow	51.64937	-1.86395	SU09	SU0994 SU0894	https://www.facebook.com/photo.php?fbid=1186356611416629&set=gm.10153907946953393&type=3
Not Provided	11/06/2016	Thetford, by River	52.41176	0.7483	TL88	TL8782 TL8883 TL8683 TL8583	https://www.facebook.com/photo.php?fbid=10206583183880279&set=gm.10154148234105330&type=3
08/06/2016	12/06/2016	Bramshill	51.34553	-0.91819	SU76	SU7461 SU7462 SU7463 SU7460 SU7563 SU7562 SU7561 SU7560 SU7663 SU7662 SU7661 SU7660	https://www.facebook.com/hampshiredragonflies/photos/a.1493532690972992/1639513673041559/?type=3
23/07/2016	23/07/2016	Bassingham	53.12934	-0.63992	SK95 SK96	SK9159 SK9160 SK9059 SK9060	https://www.facebook.com/photo.php?fbid=10210278709308820&set=gm.515869321935795&type=3
Not Provided	22/07/2016	Footscray Meadows	51.42458	0.12528	TQ47	TQ4771 TQ4871 TQ4772 TQ4872	https://www.facebook.com/AnotherEdenImages/photos/a.479792772226359/479794265559543/?type=3
31/07/2016	31/07/2016	Bassingham	53.12934	-0.63992	SK95 SK96	SK9159 SK9160 SK9059 SK9060	https://www.facebook.com/photo.php?fbid=10210356231326822&set=gm.519000561622671&type=3

Not Provided	28/06/2016	Lakenheath Fen	52.44582	0.50522	TL78	TL7286 TL7186 TL7086 TL7085 TL6986 TL6985 TL6886 TL6885 TL6785	https://www.facebook.com/photo.php?fbid=1392143497478843&set=o.170799453122627&type=3
28/06/2016	28/06/2016	Brandon Marsh	52.3771	-1.43086	SP37	SP3875 SP3975 SP3874 SP3974 SP3876 SP3976	https://www.facebook.com/photo.php?fbid=645959302226088&set=gm.819898758145735&type=3
Not Provided	04/06/2016	Gadespring Watercress, Box Moor Trust	51.74562	-0.5011	TL00	TL0306	https://www.facebook.com/DanFWildlife/photos/a.1939184429500709/1030976076988220/?type=3
05/06/2016	06/06/2016	Titchmarsh Nature Reserve	52.41276	-0.52114	TL08	TL0080 TL0081 TL0180	https://www.facebook.com/photo.php?fbid=287774368234472&set=gm.808112152657729&type=3
Not Provided	05/08/2016	Pensthorpe, Fakenham	52.82238	0.89197	TF92	TF9429 TF9529 TF9428 TF9528	https://www.facebook.com/photo.php?fbid=10210448479827134&set=gm.1085375931543486&type=3
Not Provided	24/05/2016	Church Bridge, Pocklington Canal	53.8903	-0.84727	SE74	SE7544	https://www.facebook.com/photo.php?fbid=10154108216372165&set=gm.1020524901359895&type=3
Not Provided	30/07/2016	River Marden, Calne	51.438	-2.00493	ST97 SU07	ST9870 ST9970 ST9971 SU0070	https://www.facebook.com/photo.php?fbid=10154435256357174&set=gm.661682657324393&type=3
29/07/2016	07/08/2016	Pensthorpe, Fakenham	52.82238	0.89197	TF92	TF9429 TF9529 TF9428 TF9528	https://www.facebook.com/photo.php?fbid=10210466709122855&set=gm.842124225923188&type=3
26/05/2016	26/05/2016	Hurn	50.77289	-1.82039	SZ19	SZ1297 SZ1296 SZ1295	https://www.facebook.com/hampshiredragonflies/photos/a.1493532690972992/1630613630598230/?type=3

07/06/2016	07/06/2016	Brandon Marsh	52.3771	-1.43086	SP37	SP3875 SP3975 SP3874 SP3974 SP3876 SP3976	https://www.facebook.com/photo.php?fbid=636179416537410&set=gm.808638622605082&type=3
Not Provided	06/06/2013	Irthlingborough Lakes	52.3115	-0.61075	SP97	SP9470 SP9570 SP9469 SP9569 SP9468 SP9568 SP6368 SP9468	https://www.facebook.com/photo.php?fbid=4785935888115&set=gm.10151444483002374&type=3
Not Provided	29/07/2013	Wetheral, Cumbria	54.88154	-2.83344	NY45	NY4654 NY4554 NY4555 NY4655	https://www.facebook.com/photo.php?fbid=414314792020861&set=a.239866002799075&type=3
Not Provided	30/07/2013	Houghton Mill National Trust	52.33117	-0.12009	TL27	TL2872 TL2871	https://www.facebook.com/photo.php?fbid=204746909683683&set=a.204746779683696&type=3
Not Provided	07/12/2013	Trowbridge	51.32011	-2.20852	ST85	ST8456 ST8457 ST8556 ST8557 ST8656 ST8657 ST8458 ST8558 ST8658	https://www.facebook.com/photo.php?fbid=573970172674294&set=gm.672122982838166&type=3
Not Provided	25/06/2013	Eynsford	51.36873	0.21244	TQ56	TQ5366 TQ5466 TQ5365 TQ5465 TQ5364 TQ5365	https://www.facebook.com/photo.php?fbid=4700188064180&set=a.1464496933924&type=3
Not Provided	01/06/2013	River Stour at Throop	50.7587	-1.82871	SZ19	SZ1195 SZ1295	https://www.facebook.com/Dorsetdragonflies/photos/a.372581169468176/526220400770918/?type=3

03/06/2013	03/06/2013	Marshalls Arm	53.24647	-2.52449	SJ67	SJ6472 SJ6572	https://www.facebook.com/Saltscape/photos/a.494472050600246/525957170785067/?type=3
Not Provided	25/07/2013	Moorend Spout	51.44041	-2.77113	ST47	ST4671	https://www.facebook.com/photo.php?fbid=10151730330859061&set=o.629666260379115&type=3
10/08/2013	11/08/2013	Ellesmere	52.90765	-2.89379	SJ33	SJ3934 SJ4034 SJ3935 SJ4035 SJ3936 SJ4036	https://www.facebook.com/photo.php?fbid=10153130168965644&set=a.10152252073710644&type=3
Not Provided	25/06/2013	Ham Wall Nature Reserve	51.15248	-2.77278	ST43	ST4439 ST4539 ST4639 ST4440 ST4540 ST4640	https://www.facebook.com/photo.php?fbid=10200798747125492&set=a.10200733407092032&type=3
21/06/2013	21/06/2013	Wiveton	52.94871	1.03743	TG04	TG0442 TG0443	https://www.facebook.com/photo.php?fbid=653015308046788&set=a.528318130516507&type=3
16/07/2013	17/07/2013	Sevenoakes Wildlife Reserve	51.29129	0.18197	TQ55	TQ5156 TQ5256 TQ5157 TQ5257	https://www.facebook.com/photo.php?fbid=207252212762568&set=gm.10151660502543046&type=3
14/06/2013	16/06/2013	River Ray near Swindon	51.5626	-1.80801	SU18	SU1188 SU1187 SU1287 SU1286 SU1285 SU1385 SU1384 SU1383	https://www.facebook.com/photo.php?fbid=10151697892064669&set=a.10150729864744669&type=3
Not Provided	16/06/2013	River Ray near Swindon	51.5626	-1.80801	SU18	SU1188 SU1187 SU1287 SU1286 SU1285 SU1385 SU1384 SU1383	https://www.facebook.com/photo.php?fbid=10151697892914669&set=a.10150729864744669&type=3

Not Provided	09/08/2014	Saltburn by the Sea	54.58225	-0.9736	NZ62	NZ6621 NZ6620 NZ6520 NZ6521	https://www.facebook.com/photo.php?fbid=10202345557186297&set=a.10202216107430134&type=3
Not Provided	15/05/2014	Sevenaoks Wildlife Reserve	51.29169	0.18175	TQ55	TQ5156 TQ5157 TQ5257 TQ5256	https://www.facebook.com/photo.php?fbid=305614209593925&set=gm.10152318353783046&type=3
19/06/2014	20/06/2014	Sevenaoks Wildlife Reserve	51.29169	0.18175	TQ55	TQ5156 TQ5157 TQ5257 TQ5256	https://www.facebook.com/photo.php?fbid=319168521570936&set=gm.10152411847433046&type=3
12/06/2014	13/06/2014	Bardney	53.21057	-0.32364	TF16	TF1269 TF1169	https://www.facebook.com/photo.php?fbid=10204509192114496&set=gm.244097135779683&type=3
Not Provided	11/06/2014	Wensum Park	52.64057	1.28667	TG20	TG2209	https://www.facebook.com/photo.php?fbid=508241600252&set=gm.667609376653479&type=3
Not Provided	05/07/2014	Stodmarsh National Nature Reserve (Stodge Marsh)	51.31265	1.18915	TR26	TR2161 TR2261 TR2262 TR2362	https://www.facebook.com/photo.php?fbid=739630226101973&set=gm.10152446081078046&type=3
Not Provided	24/07/2014	Launceston	50.63572	-4.36129	SX38	SX3185 SX3285 SX3385 SX3184 SX3284 SX3384 SX3183 SX3283 SX3383	https://www.facebook.com/photo.php?fbid=321534568012670&set=gm.310371385789309&type=3
Not Provided	28/06/2014	Brandon Marsh	52.3771	-1.43086	SP37	SP3875 SP3975 SP3874 SP3974 SP3876 SP3976	https://www.facebook.com/treethatfellinthewoods/photos/a.262978500553128/271974372986874/?type=3

Not Provided	04/01/2014	Lound, Nottinghamshire	53.37065	-0.96278	SK68	SK6986 SK6985 SK6885 SK6886 SK6887 SK6987	https://www.facebook.com/photo.php?fbid=783094058371738&set=a.403595149654966&type=3
12/06/2014	12/06/2014	Bardney	53.21057	-0.32364	TF16	TF1269 TF1169	https://www.facebook.com/photo.php?fbid=10204499464551313&set=gm.243828422473221&type=3
Not Provided	11/05/2014	River Wear, Durham	54.77794	-1.57776	NZ24	NZ2944 NZ2844 NZ2843 NZ2743 NZ2742 NZ2741 NZ2842 NZ2841	https://www.facebook.com/photo.php?fbid=10201567930392465&set=gm.445065855629029&type=3
17/07/2014	17/07/2014	Twickenham Riverside	51.44746	-0.31213	TQ17	TQ1675 TQ1774 TQ1773 TQ1673 TQ1672 TQ1671	https://www.facebook.com/photo.php?fbid=10154377630025075&set=gm.275903992614117&type=3
Not Provided	09/06/2014	Needham Market Lake	52.15167	1.05855	TM05	TM0954	https://www.facebook.com/photo.php?fbid=516260078501528&set=a.157240327736840&type=3
12/06/2014	13/06/2014	Bardney	53.21057	-0.32364	TF16	TF1269 TF1169	https://www.facebook.com/photo.php?fbid=10204508158488656&set=gm.244066832449380&type=3
Not Provided	05/06/2014	Snakeholme Pit	53.23009	-0.33022	TF17	TF1171	https://www.facebook.com/photo.php?fbid=1428766800722199&set=gm.456583324477282&type=3
Not Provided	03/07/2014	Fordwich	51.29524	1.12597	TR15	TR1759 TR1859 TR1760 TR1860	https://www.facebook.com/photo.php?fbid=775429865822672&set=gm.1015244145189804 6&type=3

Not Provided	21/08/2014	River Waveney, Scole	52.36147	1.15613	TM17	TM1478 TM1578	https://www.facebook.com/trevorsdigitalphotography/photos/a.1462789323990801/1462801350656265/?type=3
Not Provided	25/06/2014	University of East Anglia	52.6216	1.23854	TG10	TG1907 TG1807	https://www.facebook.com/photo.php?fbid=722734187783717&set=a.103371706386638&type=3
Not Provided	26/05/2014	Dearne Valley Park, Barnsley	53.55752	-1.45646	SE30	SE3606 SE3506 SE3607	https://www.facebook.com/photo.php?fbid=668697436517925&set=gm.643041809108208&type=3
Not Provided	28/07/2014	Middleton Lakes	52.59403	-1.70086	SP19 SP29 SK20	SP1998 SP2098 SP2099 SK2000	https://www.facebook.com/photo.php?fbid=741274272585604&set=gm.570804673030178&type=3
Not Provided	10/06/2014	Priory Country Park, Bedford	52.12992	-0.43099	TL04	TL0749 TL0748 TL0648 TL0649	https://www.facebook.com/RobertFeltonPhotography/photos/a.201983696600742/495483350584107/?type=3
Not Provided	03/08/2014	Upton Warren	52.30507	-2.10401	SO96	SO9367 SO9368 SO9267 SO9268	https://www.facebook.com/louisemorrisimages/photos/a.631475016865148/825101687502479/?type=3
Not Provided	16/05/2014	Sankey Valley Park, along canal	53.3998	-2.61627	SJ58	SJ5989 SJ5988	https://www.facebook.com/photo.php?fbid=587436928030480&set=o.110886079024465&type=3
Not Provided	05/07/2014	Stodmarsh National Nature Reserve (Stodge Marsh)	51.31265	1.18915	TR26	TR2161 TR2261 TR2262 TR2362	https://www.facebook.com/RWFordPhotography/photos/a.724435810957019/733170256750241/?type=3
Not Provided	24/06/2014	Rushcliffe Country Park	52.88303	-1.14896	SK53	SK5732 SK5832 SK5731 SK5631 SK5632	https://www.facebook.com/photo.php?fbid=10204189762454343&set=gm.672041332851248&type=3

06/06/2014	08/06/2014	Attenborough nature reserve	52.90397	-1.22525	SK53	SK5033 SK5133 SK5233 SK5134 SK5234 SK5235 SK5335	https://www.facebook.com/photo.php?fbid=10204059777524801&set=gm.664843343571047&type=3
18/05/2014	18/05/2014	Whaddon	52.09698	-0.03208	TL34	TL3446 TL3546	https://www.facebook.com/photo.php?fbid=657317317672912&set=gm.448272868641661&type=3
06/08/2014	06/08/2014	Hexthorpe, River Don	53.51736	-1.16327	SE50	SE5603 SE5602 SE5502 SE5501	https://www.facebook.com/photo.php?fbid=10201440629936293&set=gm.486979421437672&type=3
Not Provided	19/07/2014	Stodmarsh National Nature Reserve (Stodge Marsh)	51.31265	1.18915	TR26	TR2161 TR2261 TR2262 TR2362	https://www.facebook.com/photo.php?fbid=10202380790352866&set=gm.10152474623618046&type=3
Not Provided	24/06/2014	Ribchester	53.81383	-2.53277	SD63	SD6535 SD6435 SD6436 SD6536	https://www.facebook.com/photo.php?fbid=10201891713487317&set=a.1301301345455&type=3
Not Provided	27/07/2014	Greylake Nature Reserve	51.11066	-2.86609	ST33	ST3934 ST3935	https://www.facebook.com/photo.php?fbid=10202405679165150&set=a.10202405678725139&type=3
Not Provided	17/05/2014	Hurn	50.77289	-1.82039	SZ19	SZ1297 SZ1296 SZ1295	https://www.facebook.com/photo.php?fbid=10152515389766929&set=a.10150103476301929&type=3
Not Provided	15/07/2014	Warwick Hall	54.90444	-2.8332	NY45	NY4656	https://www.facebook.com/CarlisleNats/photos/a.406728006427/10152104734701428/?type=3
Not Provided	15/07/2014	Warwick Hall	54.90444	-2.8332	NY45	NY4656	https://www.facebook.com/CarlisleNats/photos/a.406728006427/10152104734176428/?type=3

13/07/2014	19/07/2014	Westport Lake	53.04825	-2.21581	SJ85 SJ84	SJ8550 SJ8549	https://www.facebook.com/photo.php?fbid=688643777869338&set=o.66801316795&type=3
18/05/2014	19/05/2014	Stopham Bridge, West Sussex	50.95566	-0.53546	TQ01	TQ0318 TQ0218	https://www.facebook.com/photo.php?fbid=10203505101404802&set=o.176490629078699&type=3
Not Provided	31/12/2014	River Derwent, Howardian Hills	54.08992	-0.87507	SE76	SE7669 SE7666 SE7566 SE7567 SE7467 SE7367 SE7366 SE7365 SE7465 SE7464 SE7463 SE7363	https://www.facebook.com/129557403821143/photos/a.649395668503978/649399391836939/?type=3
Not Provided	27/12/2014	Woolston Eyes Nature Reserve	53.39084	-2.51926	SJ68	SJ6388 SJ6488 SJ6487 SJ6588 SJ6587 SJ6688 SJ6788 SJ6689 SJ6789	https://www.facebook.com/photo.php?fbid=935135393171205&set=a.935134723171272&type=3
13/05/2014	13/05/2014	Earsham	52.45052	1.41595	TM38	TM3289 TM3288	https://www.facebook.com/photo.php?fbid=10201031279544355&set=o.615032165232886&type=3
Not Provided	01/08/2014	Whitwell, Isle of Wight	50.59784	-1.26452	SZ57	SZ5277 SZ5177 SZ5278 SZ5178	https://www.facebook.com/photo.php?fbid=616441475142308&set=a.398776000242191&type=3
Not Provided	14/12/2014	Piddle Brook Meadows	52.17419	-2.055	SO95	SO9652	https://www.facebook.com/WychavonActionGroup/photos/a.1452217441714259/1510579715878031/?type=3

Not Provided	05/06/2014	Upper Wolvercote, by Oxford Canal	51.78383	-1.28432	SP40 SP41	SP4910 SP4909	https://www.facebook.com/photo.php?fbid=674259045961245&set=a.177815675605587&type=3
Not Provided	24/07/2014	Lach Dennis, canal side	53.24014	-2.47682	SJ67	SJ6872 SJ6871 SJ6870	https://www.facebook.com/photo.php?fbid=556878997771746&set=a.127668967359420&type=3
Not Provided	10/06/2014	Winnall Moors	51.07625	-1.30139	SU42 SU43	SU4829 SU4830 SU4930 SU4831 SU4932	https://www.facebook.com/153132638110668/photos/a.652681138155813/652702504820343/?type=3
18/05/2014	05/09/2014	River Itchen, Winchester	51.06161	-1.30769	SU42 SU43	SU4931 SU4930 SU4832 SU4830 SU4829 SU4828 SU4827 SU4727	https://www.facebook.com/photo.php?fbid=10152646375505664&set=a.4711400663&type=3
Not Provided	11/08/2014	Jubilee Lake	51.55318	-1.90273	SU08	SU0683	https://www.facebook.com/234044973350686/photos/a.307695735985609/666886020066577/?type=3
Not Provided	31/05/2014	Woolston Eyes Nature Reserve	53.39084	-2.51926	SJ68	SJ6388 SJ6488 SJ6487 SJ6588 SJ6587 SJ6688 SJ6788 SJ6689 SJ6789	https://www.facebook.com/photo.php?fbid=807748922576520&set=a.524534987564583&type=3
20/07/2014	20/07/2014	Rooksury Mill	51.19859	-1.49248	SU34	SU3544	https://www.facebook.com/photo.php?fbid=10152540458570664&set=gm.10152571470866031&type=3
15/05/2014	16/05/2014	Wolseley Centre	52.7794	-1.96544	SK02	SK0220	https://www.facebook.com/WolseleyCentre/photos/a.265959350124151/677133845673364/?type=3

Not Provided	21/05/2014	Sankey Valley Park	53.3998	-2.61627	SJ58	SJ5989 SJ5988	https://www.facebook.com/photo.php?fbid=589875841119922&set=o.555898354447451&type=3
Not Provided	28/05/2014	Darley and Nutwood Local Nature Reserve	52.94618	-1.47369	SK33	SK3538 SK3539	https://www.facebook.com/darleyabbeysociety/photos/a.484711578222087/885067791519795/?type=3
Aug-14	16/11/2014	Ravensroost, Wiltshire	51.59375	-1.96162	SU08	SU0288 SU0287	https://www.facebook.com/376744642449101/photos/a.376779692445596/608588625931367/?type=3
Not Provided	17/07/2014	Mire Loch	55.91185	-2.14159	NT96	NT9168	https://www.facebook.com/213846425386017/photos/a.227996447304348/543257485778241/?type=3
Not Provided	09/08/2014	Snodland Brooks	51.32437	0.4529	TQ76	TQ7061 TQ7161 TQ7060 TQ7160	https://www.facebook.com/333265860142783/photos/a.333285953474107/492420010894033/?type=3
Not Provided	26/06/2014	Nightingale Wood	51.59279	-1.69893	SU28	SU2188 SU2088 SU2187	https://www.facebook.com/photo.php?fbid=486026261541131&set=a.114391748704586&type=3
May-14	21/05/2014	Barcombe Mill	50.91545	0.0394	TQ41	TQ4314	https://www.facebook.com/photo.php?fbid=10152899843789126&set=o.95994239920&type=3
12/06/2014	14/06/2014	Bardney	53.21057	-0.32364	TF16	TF1269 TF1169	https://www.facebook.com/photo.php?fbid=10204513354258547&set=gm.244217259101004&type=3
12/06/2014	14/06/2014	Bardney	53.21057	-0.32364	TF16	TF1269 TF1169	https://www.facebook.com/photo.php?fbid=10204513354298548&set=gm.244217259101004&type=3
12/06/2014	14/06/2014	Bardney	53.21057	-0.32364	TF16	TF1269 TF1169	https://www.facebook.com/photo.php?fbid=10204513354378550&set=gm.244217259101004&type=3

Not Provided	24/07/2014	Lach Dennis, canal side	53.24014	-2.47682	SJ67	SJ6872 SJ6871 SJ6870	https://www.facebook.com/photo.php?fbid=556878924438420&set=a.127668967359420&type=3
20/06/2014	20/06/2014	Stodmarsh National Nature Reserve (Stodge Marsh)	51.31265	1.18915	TR26	TR2161 TR2261 TR2262 TR2362	https://www.facebook.com/photo.php?fbid=10202787305418357&set=o.519061651519845&type=3
Not Provided	27/07/2014	RSPB Strumpshaw Fen	52.60337	1.45373	TG30	TG3405 TG3406 TG3305 TG3306 TG3307	https://www.facebook.com/photo.php?fbid=1437089779906518&set=gm.690639617683788&type=3
09/06/2014	09/06/2014	Cricklade, by Thames	51.64454	-1.85545	SU19 SU09	SU1093 SU1094 SU0994	https://www.facebook.com/photo.php?fbid=809851352361415&set=a.102869356392955&type=3
Not Provided	30/05/2014	River Aire near Skipton	53.94892	-2.03391	SD95	SD9651 SD9650 SD9750 SD9850	https://www.facebook.com/photo.php?fbid=474714155992688&set=gm.644914802254242&type=3
Not Provided	23/07/2014	RSPB Old Moor	53.51784	-1.35268	SE40	SE4202 SE4302	https://www.facebook.com/OldMoorRSPB/photos/a.243290042512377/324429431065104/?type=3
20/05/2014	20/05/2014	Brockoles, River Ribble	53.77178	-2.62155	SD53	SD5830 SD5930 SD5831	https://www.facebook.com/photo.php?fbid=704211619640445&set=a.496477060413903&type=3
31/05/2014	01/06/2014	Nightingale Wood	51.59279	-1.69893	SU28	SU2188 SU2088 SU2187	https://www.facebook.com/photo.php?fbid=10152516770309343&set=a.10152189237049343&type=3
20/08/2014	21/08/2014	Millers Pond, Hardwick Hall	53.17328	-1.32201	SK46	SK4564	https://www.facebook.com/photo.php?fbid=4456991759027&set=o.136917996370272&type=3
Not Provided	14/08/2014	Wolston Eyes Nature Reserve, No.1 bed	53.39522	-2.49508	SJ68	SJ6688 SJ6788 SJ6689 SJ6789	https://www.facebook.com/photo.php?fbid=853366174681461&set=o.690391564350150&type=3

Not Provided	15/07/2014	Brockdale Nature Reserve on River Went	53.64944	-1.24068	SE51	SE5117 SE5116 SE5017 SE4917	https://www.facebook.com/photo.php?fbid=705910269464776&set=gm.1651231368435632&type=3
Not Provided	15/07/2014	Brockdale Nature Reserve on River Went	53.64944	-1.24068	SE51	SE5117 SE5116 SE5017 SE4917	https://www.facebook.com/photo.php?fbid=705910389464764&set=gm.1651231368435632&type=3
Not Provided	23/07/2014	Pullborough Brooks Nature Reserve	50.94406	-0.50385	TQ01	TQ0516 TQ0416 TQ0517	https://www.facebook.com/photo.php?fbid=253596148097408&set=gm.479786462156968&type=3
19/05/2014	20/05/2014	Lunt Meadows	53.51943	-2.988	SD30	SD3402 SD3502 SD3403	https://www.facebook.com/photo.php?fbid=1435279006743268&set=a.1380749985529504&type=3
Jun-14	23/09/2014	River Dee, Chester	53.18366	-2.89409	SJ46	SJ3765 SJ3865 SJ3966 SJ3965 SJ4065 SJ4166 SJ4165 SJ4164	https://www.facebook.com/photo.php?fbid=317484528424594&set=gm.513975388738075&type=3
22/06/2014	22/06/2014	Shackerstone	52.65744	-1.44957	SK30	SK3706	https://www.facebook.com/photo.php?fbid=10203266447415180&set=gm.464844676984480&type=3
22/06/2014	22/06/2014	Shackerstone	52.65744	-1.44957	SK30	SK3706	https://www.facebook.com/photo.php?fbid=10203266444135098&set=gm.464844676984480&type=3
Not Provided	27/06/2015	Cuerden Valley Park	53.70895	-2.66558	SD52	SD5624 SD5623 SD5524 SD5523	https://www.facebook.com/photo.php?fbid=966126880085369&set=a.773431719354887&type=3
Not Provided	22/07/2015	High Eske Nature Reserve	53.88189	-0.40215	TA04	TA0543 TA0443 TA0444 TA0544	https://www.facebook.com/photo.php?fbid=861845273910219&set=gm.943263529029927&type=3

Not Provided	25/06/2015	Grove Ferry near Canterbury	51.32174	1.21222	TR26	TR2462 TR2463 TR2363 TR2362	https://www.facebook.com/photo.php?fbid=10153428096322812&set=gm.573299632813161&type=3
Not Provided	12/08/2015	Calne	51.43784	-2.00475	ST97	ST9872 ST9871 ST9870 ST9869 ST9972 ST9971 ST9970 ST9969 SU0072 SU0071 SU0070 SU0069	https://www.facebook.com/photo.php?fbid=10153573094337174&set=gm.517161495109844&type=3
08/08/2015	08/08/2015	Stour at Grove Ferry	51.3206	1.21613	TR26	TR2262 TR2363 TR2462 TR2463	https://www.facebook.com/photo.php?fbid=948805925184401&set=gm.592436804232777&type=3
Not Provided	30/07/2015	Micklemere Nature Reserve	52.29171	0.84015	TL96	TL9369	https://www.facebook.com/photo.php?fbid=10153036429716733&set=gm.656480207820925&type=3
Not Provided	02/08/2015	Lakenheath Fen	52.44582	0.50522	TL78	TL7286 TL7186 TL7086 TL7085 TL6986 TL6985 TL6886 TL6885 TL6785	https://www.facebook.com/photo.php?fbid=10153450310963376&set=gm.758490314262424&type=3
03/06/2015	03/06/2015	Kirkby Gravel Pits, Lincolnshire Wildlife Trust	53.13369	-0.1501	TF26	TF2361 TF2360	https://www.facebook.com/photo.php?fbid=1587131728219038&set=gm.384277928428269&type=3
Not Provided	04/08/2015	Low Hall Nature Reserve, near Hindley	53.52452	-2.58741	SD60	SD6003 SD6103	https://www.facebook.com/photo.php?fbid=1661059414140287&set=gm.1875495886008836&type=3

Not Provided	04/06/2015	Riverside Valley Park, Exeter	50.70985	-3.52116	SX99	SX9291 SX9290	https://www.facebook.com/Devonwildlife/photos/a.188231861212213/850151835020209/?type=3
29/09/2015	29/09/2015	Wigan dry dock	53.54033	-2.63786	SD50	SD5705	https://www.facebook.com/photo.php?fbid=1633928353547343&set=a.1550212868585559&type=3
11/06/2015	18/06/2015	Needham Lakes, Suffolk	52.15131	1.05901	TM05	TM0954	https://www.facebook.com/photo.php?fbid=1158664484149351&set=a.202608486421627&type=3
22/05/2015	26/11/2015	Staveley, Derbyshire	53.26584	-1.35228	SK47	SK4172 SK4173 SK4174 SK4175 SK4272 SK4273 SK4274 SK4275 SK4372 SK4373 SK4374 SK4375 SK4472 SK4473 SK4474 SK4475	https://www.facebook.com/photo.php?fbid=1659021514368816&set=gm.637434696399654&type=3
Not Provided	01/07/2015	Whaddon	52.09698	-0.03208	TL34	TL3446 TL3546	https://www.facebook.com/photo.php?fbid=873306629407312&set=gm.877398795641808&type=3
22/05/2015	22/05/2015	Lakenheath Fen	52.44582	0.50522	TL78	TL7286 TL7186 TL7086 TL7085 TL6986 TL6985 TL6886 TL6885 TL6785	https://www.facebook.com/photo.php?fbid=10204647291877299&set=gm.821714927883887&type=3
Not Provided	03/07/2015	Dabton Loch, near Thornhill	55.25329	-3.7673	NX89	NX8796	https://www.facebook.com/photo.php?fbid=10207539512949563&set=gm.10153109865387

							950&type=3
30/05/2015	30/05/2015	Whaddon	52.09698	-0.03208	TL34	TL3446 TL3546	https://www.facebook.com/photo.php?fbid=852831611454814&set=gm.862942103754144&type=3
Not Provided	09/07/2015	Aylestone Meadows, Leicester	52.61116	-1.15019	SK50	SK5701 SK5702	https://www.facebook.com/ColinStanleyPhotographyArtwork/photos/a.511740572263322/706093512828026/?type=3
22/05/2015	22/05/2015	Attenborough nature reserve	52.90397	-1.22525	SK53	SK5033 SK5133 SK5233 SK5134 SK5234 SK5235 SK5335	https://www.facebook.com/313752532013173/photos/a.367223223332770/843383225716765/?type=3
Not Provided	12/07/2015	Westenhanger	51.09093	1.03732	TR13	TR1237 TR1236	https://www.facebook.com/photo.php?fbid=10153490742082840&set=a.10153470377447840&type=3
15/07/2015	16/07/2015	Preston Marshes	51.29843	1.2014	TR26	TR2360	https://www.facebook.com/photo.php?fbid=10153221374231144&set=gm.10153351853698046&type=3
Not Provided	08/06/2015	Boilton Wood	53.77511	-2.64268	SD53	SD5730 SD5731 SD5831	https://www.facebook.com/121232957894928/photos/a.205031052848451/1012210425463839/?type=3
Not Provided	23/05/2015	River Foss, upstream if outer ring road	54.00682	-1.06219	SE65	SE6157	https://www.facebook.com/photo.php?fbid=974350329272320&set=gm.826209017458152&type=3
13/05/2015	13/05/2015	Warfield, Berkshire	51.44229	-0.73441	SU87	SU8772 SU8872	https://www.facebook.com/photo.php?fbid=10153267681169522&set=gm.633098873492392&type=3
Not Provided	23/06/2015	Hall Place	51.4463	0.16188	TQ57	TQ5074	https://www.facebook.com/photo.php?fbid=10206511144838558&set=gm.725857087524935&type=3

30/05/2015	30/05/2015	Whaddon	52.09698	-0.03208	TL34	TL3446 TL3546	https://www.facebook.com/photo.php?fbid=852833044788004&set=gm.641495282652751&type=3
Not Provided	30/06/2015	Kingsmead Field	51.2868	1.08393	TR15	TR1558	https://www.facebook.com/photo.php?fbid=10153188823045186&set=gm.910023272397665&type=3
Aug-15	05/08/2015	Roman Camp Hotel, Aylmerton	52.9182	1.24713	TG14	TG1840	https://www.facebook.com/photo.php?fbid=10207278567233079&set=a.10204389370884976&type=3
Not Provided	21/06/2015	Wilden Marsh	52.35964	-2.25917	SO87	SO8273	https://www.facebook.com/thewildenmarshblog/photos/o.94978880994/568842179925512/?type=3
Not Provided	18/08/2015	Symonds Yat	51.84729	-2.63748	SO51	SO5516 SO5517 SO5518 SO5616 SO5617 SO5618	https://www.facebook.com/HayleyIrelandPhotography/photos/a.1613832262223713/1614860028787603/?type=3
16/06/2015	16/06/2015	Swafield, canal	52.83677	1.39264	TG23	TG2832 TG2831	https://www.facebook.com/photo.php?fbid=1021065731236908&set=o.436444736462985&type=3
Not Provided	11/07/2015	Aylestone Meadows, Leicester	52.61116	-1.15019	SK50	SK5701 SK5702	https://www.facebook.com/ColinStanleyPhotographyArtwork/photos/a.511740572263322/702188459885198/?type=3

Not Provided	12/10/2015	Crowhurst and Central Valley SSSI	50.86429	0.50828	TQ71 TQ70	TQ7413 TQ7412 TQ7411 TQ7513 TQ7512 TQ7511 TQ7613 TQ7612 TQ7611 TQ7510 TQ7610 TQ7710 TQ7709	https://www.facebook.com/CombeValley/photos/a.876260355793217/886694141416505/?type=3
Not Provided	06/06/2015	Barcombe, River Ouse	50.9151	0.03264	TQ41	TQ4214 TQ4314 TQ4313 TQ4315	https://www.facebook.com/photo.php?fbid=807964299323357&set=a.398776000242191&type=3

Table A.3. Banded Demoiselle (Calopteryx splendens) occurrences extracted from Flickr using the Flickr API.

Date Recorded	Date Posted	Location Description	Latitude	Longitude	Website
22/06/2021	23/06/2021	Fulford	53.928286	-1.082614	https://www.flickr.com/photos/smithandjones/51267817180/
16/06/2021	04/07/2021	Three Rivers	51.668868	-0.502506	https://www.flickr.com/photos/crashcalloway/51288564906/
08/06/2013	Not Provided	Hartlebury	52.334001	-2.251853	https://www.flickr.com/photos/63588774@N06/8997420655
23/07/2021	17/08/2021	Bird End	52.525632	-1.962712	https://www.flickr.com/photos/michaeljh/51383941916/
14/06/2013	Not Provided	Swindon	51.596879	-1.688424	https://www.flickr.com/photos/15121707@N00/9057294534
15/06/2021	01/09/2021	Brinsworth	53.424135	-1.368092	https://www.flickr.com/photos/petermit2/51416599853/
15/06/2021	01/09/2021	Brinsworth	53.424135	-1.368092	https://www.flickr.com/photos/petermit2/51415587837/
15/06/2021	01/09/2021	Brinsworth	53.424135	-1.368092	https://www.flickr.com/photos/petermit2/51416599783/
19/07/2021	04/02/2022	South Derbyshire District	52.882533	-1.335160	https://www.flickr.com/photos/pjfulford/51860798053/
02/07/2021	11/01/2022	South Derbyshire District	52.882222	-1.334946	https://www.flickr.com/photos/pjfulford/51812387932/
31/05/2017	Not Provided	Attenborough	52.902202	-1.232314	https://www.flickr.com/photos/8404368@N04/35048578541
16/06/2015	Not Provided	Colwick Park	52.952361	-1.092066	https://www.flickr.com/photos/43301211@N03/20926709911
22/06/2021	07/07/2021	Bishopthorpe	53.927156	-1.086573	https://www.flickr.com/photos/smithandjones/51295831438/
30/05/2021	10/06/2021	Leighton-Linslade	51.928900	-0.671806	https://www.flickr.com/photos/steveinleightonsphotos/51238300019/
17/06/2021	02/08/2021	Felmersham	52.212772	-0.554606	https://www.flickr.com/photos/oandrews/51353904794/
29/07/2014	Not Provided	Old Woking	51.284722	-0.550045	https://www.flickr.com/photos/86182676@N00/

					14599370260
09/05/2020	Not Provided	Send	51.296866	-0.52479	https://www.flickr.com/photos/86182676@N00/49879147866
02/07/2018	Not Provided	Surrey	51.386914	-0.429453	https://www.flickr.com/photos/86182676@N00/43223098541
05/06/2021	05/06/2021	Richmond upon Thames	51.408361	-0.317798	https://www.flickr.com/photos/98609459@N08/51225436402/
04/08/2021	15/08/2021	St. Edmundsbury District	52.306130	0.640769	https://www.flickr.com/photos/bobchappell/51378365567/
09/06/2020	23/07/2021	St. Edmundsbury District	52.305147	0.648977	https://www.flickr.com/photos/bobchappell/51329657473/
09/06/2021	15/08/2021	South Norfolk District	52.628297	1.234521	https://www.flickr.com/photos/43688219@N00/51379363656/
09/06/2021	15/08/2021	South Norfolk District	52.628297	1.234521	https://www.flickr.com/photos/43688219@N00/51378601817/
10/08/2021	10/08/2021	West Devon District	50.850600	-4.095106	https://www.flickr.com/photos/aneyetothehills/51368938027/
20/06/2021	22/06/2021	Exeter District	50.712289	-3.521375	https://www.flickr.com/photos/170969385@N05/51263755093/
31/07/2015	31/07/2015	Cardiff	51.517205	-3.246012	https://www.flickr.com/photos/87670724@N08/20184449251
25/07/2016	12/04/2021	Llanarth	51.773719	-2.929927	https://www.flickr.com/photos/hoppy1951/51112586260/
02/06/2021	03/06/2021	Ellesmere Urban	52.892722	-2.913348	https://www.flickr.com/photos/47046427@N03/51222416806/
01/06/2021	03/06/2021	West Lancashire District	53.616789	-2.871830	https://www.flickr.com/photos/31693460@N06/51222119532/
22/07/2021	23/07/2021	Rixton-with-Glazebrook	53.407604	-2.476985	https://www.flickr.com/photos/petercollier65/51329706253/
22/07/2021	23/07/2021	Rixton-with-Glazebrook	53.407604	-2.476985	https://www.flickr.com/photos/petercollier65/51328760927/
04/08/2021	05/08/2021	Sandbach	53.131072	-2.392632	https://www.flickr.com/photos/wistaston/51358773060/
14/06/2021	26/06/2021	Worcester District	52.183297	-2.222832	https://www.flickr.com/photos/davea2007/

					51271187941/
23/07/2021	06/09/2021	Bird End	52.525184	-1.963870	https://www.flickr.com/photos/michaeljh/51428902682/
23/07/2021	13/08/2021	Bird End	52.525445	-1.963119	https://www.flickr.com/photos/michaeljh/51375563263/
23/07/2021	04/09/2021	Bird End	52.526051	-1.962840	https://www.flickr.com/photos/michaeljh/51423809354/
23/07/2021	26/08/2021	Bird End	52.525632	-1.962712	https://www.flickr.com/photos/michaeljh/51402907462/
23/07/2021	18/08/2021	Bird End	52.525632	-1.962712	https://www.flickr.com/photos/michaeljh/51385952726/
04/05/2011	Not Provided	Leigh	51.638137	-1.914901	https://www.flickr.com/photos/15121707@N00/5688113955
24/07/2017	Not Provided	Bournemouth	50.767034	-1.87569	https://www.flickr.com/photos/16054928@N07/36193266241
29/05/2021	04/06/2021	Durnford	51.135638	-1.814020	https://www.flickr.com/photos/15543444@N00/51224531033/
14/06/2013	Not Provided	Swindon	51.596879	-1.688424	https://www.flickr.com/photos/15121707@N00/9055075385
15/06/2021	15/06/2021	South Derbyshire District	52.879788	-1.629806	https://www.flickr.com/photos/141463739@N03/51249230898/
05/06/2010	Not Provided	Swinford	51.775501	-1.352283	https://flickr.com/photos/62308688@N00/4672912684
26/06/2021	31/01/2022	South Derbyshire District	52.882915	-1.335321	https://www.flickr.com/photos/pjfulford/51853642524/
14/07/2021	12/01/2022	South Derbyshire District	52.882397	-1.334978	https://www.flickr.com/photos/pjfulford/51814582642/
03/06/2018	Not Provided	Toton	52.906361	-1.268395	https://www.flickr.com/photos/rorysmith/49018989638/in/dateposted/
07/06/2021	10/06/2021	South Oxfordshire District	51.653282	-1.210792	https://www.flickr.com/photos/bruce-clarke/51237078976/
18/05/2014	Not Provided	West Bridgford	52.942741	-1.129446	https://www.flickr.com/photos/42985099@N03/14235159043
05/06/2015	Not Provided	Colwick Park	52.949361	-1.086831	https://www.flickr.com/photos/43301211@N03/

					18459223199
22/06/2021	23/06/2021	Fulford	53.928343	-1.082260	https://www.flickr.com/photos/smithandjones/51266458755/
29/05/2020	04/03/2021	South Oxfordshire District	51.541824	-0.898492	https://www.flickr.com/photos/moblynn/51002398196/
08/06/2021	06/07/2021	Wellingborough District	52.260186	-0.731470	https://www.flickr.com/photos/gillybooze/51293647611/
07/06/2021	11/06/2021	Wellingborough District	52.262527	-0.728856	https://www.flickr.com/photos/gillybooze/51240615945/
30/05/2021	10/06/2021	Leighton-Linslade	51.928900	-0.671806	https://www.flickr.com/photos/steveinleightonsphotos/51238586080/
17/06/2021	03/08/2021	Felmersham	52.215275	-0.550887	https://www.flickr.com/photos/oandrews/51356052899/
02/06/2021	07/06/2021	Colnbrook with Poyle	51.475141	-0.512623	https://www.flickr.com/photos/coptercrazy/51230127042/
02/06/2021	07/06/2021	Colnbrook with Poyle	51.475141	-0.512623	https://www.flickr.com/photos/coptercrazy/51231045703/
02/06/2021	07/06/2021	Colnbrook with Poyle	51.475141	-0.512623	https://www.flickr.com/photos/coptercrazy/51230836746/
02/06/2021	07/06/2021	Colnbrook with Poyle	51.475141	-0.512623	https://www.flickr.com/photos/coptercrazy/51230126632/
16/06/2021	03/07/2021	Three Rivers	51.668868	-0.502506	https://www.flickr.com/photos/crashcalloway/51287285618/
17/05/2014	Not Provided	Hatton	51.460709	-0.430612	https://www.flickr.com/photos/44691276@N06/14096577109
23/06/2021	06/07/2021	Tempsford	52.171494	-0.303650	https://www.flickr.com/photos/jccurd/51294214039/
05/07/2021	06/07/2021	Biggleswade	52.077303	-0.271615	https://www.flickr.com/photos/27045884@N05/51293729898/
06/08/2021	09/08/2021	Haringey	51.587780	-0.054612	https://www.flickr.com/photos/47046427@N03/51368248895/
26/07/2021	05/08/2021	St. Edmundsbury District	52.309070	0.638143	https://www.flickr.com/photos/pstani/51358288447/
15/06/2020	17/06/2021	St. Edmundsbury District	52.305033	0.648872	https://www.flickr.com/photos/bobchappell/

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15/06/2020	13/02/2022	St. Edmundsbury District	52.305055	0.648894	https://www.flickr.com/photos/bobchappell/51877666627/
15/06/2020	30/01/2022	St. Edmundsbury District	52.305047	0.648936	https://www.flickr.com/photos/bobchappell/51851507548/
05/07/2021	07/07/2021	North Norfolk District	52.942522	1.140936	https://www.flickr.com/photos/65272192@N02/51296649828/
05/06/2010	Not Provided	Leicester	52.629012	-1.143618	https://flickr.com/photos/73684697@N00/4674042031
22/06/2010	Not Provided	Swinford	51.775687	-1.357884	https://flickr.com/photos/62308688@N00/4843727393
01/07/2019	Not Provided	Binley	52.397984	-1.462748	https://www.flickr.com/photos/15181848@N02/48170108381
01/07/2019	Not Provided	Binley	52.397984	-1.462748	https://www.flickr.com/photos/15181848@N02/48170107851
30/07/2016	01/08/2016	St. Levan	50.040115	-5.678021	https://www.flickr.com/photos/julianhodgson/28665168056/
15/08/2018	15/08/2018	Helston	50.102102	-5.281050	https://www.flickr.com/photos/16498755@N07/43336288234/
15/06/2011	13/02/2013	Shortlanesend	50.286268	-5.095252	https://www.flickr.com/photos/raggedjack/8471950154/
24/05/2019	07/06/2019	Torridge District	50.879644	-4.432137	https://www.flickr.com/photos/julianbaird/48017786008/
24/05/2019	05/06/2019	Torridge District	50.879644	-4.432137	https://www.flickr.com/photos/julianbaird/48006659318/
01/06/2020	01/06/2020	West Devon District	50.850175	-4.095337	https://www.flickr.com/photos/aneyetothehills/49959921353/
26/06/2018	26/06/2018	Dolton	50.895786	-4.057523	https://www.flickr.com/photos/aneyetothehills/42305660114/
04/07/2015	04/07/2015	Carmarthenshire County	51.882002	-4.014387	https://www.flickr.com/photos/lilo_lil/19226488050/
01/08/2013	01/08/2013	Ceredigion County	52.252442	-3.923953	https://www.flickr.com/photos/hoppy1951/9419241068/
28/08/2012	29/08/2012	Merthyr Mawr	51.482150	-3.605708	https://www.flickr.com/photos/bob_hopkins/

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29/05/2018	01/06/2018	Devon	50.712506	-3.459641	https://www.flickr.com/photos/hickatee/41600887685/
13/06/2017	13/06/2017		53.272245	-3.452968	https://www.flickr.com/photos/96747463@N05/35246782866/
15/06/2018	19/06/2018	Stonyford	50.852680	-3.382802	https://www.flickr.com/photos/hickatee/42190408154/
11/07/2020	28/11/2020	Mid Devon District	50.851948	-3.382415	https://www.flickr.com/photos/hickatee/50655912021/
13/07/2018	15/08/2018	Stonyford	50.855065	-3.382051	https://www.flickr.com/photos/hickatee/29119700947/
09/07/2019	30/07/2019	East Devon District	50.663991	-3.311917	https://www.flickr.com/photos/68607739@N06/48416793697/
09/07/2019	30/07/2019	East Devon District	50.663991	-3.311917	https://www.flickr.com/photos/68607739@N06/48416638121/
06/06/2018	21/06/2018	Kersbrook	50.643836	-3.310467	https://www.flickr.com/photos/68607739@N06/29064517418/
19/06/2017	20/06/2017	East Devon District	50.651336	-3.305920	https://www.flickr.com/photos/68607739@N06/35391401476/
13/07/2013	14/07/2013	Montgomery	52.575052	-3.187258	https://www.flickr.com/photos/40011/9283147408/
13/07/2013	14/07/2013	Montgomery	52.575052	-3.187258	https://www.flickr.com/photos/40011/9280366597/
13/07/2013	14/07/2013	Montgomery	52.575052	-3.187258	https://www.flickr.com/photos/40011/9283144466/
13/07/2013	14/07/2013	Montgomery	52.575052	-3.187258	https://www.flickr.com/photos/40011/9280362879/
13/07/2013	14/07/2013	Montgomery	52.575052	-3.187258	https://www.flickr.com/photos/40011/9280361065/
13/07/2013	14/07/2013	Montgomery	52.575052	-3.187258	https://www.flickr.com/photos/40011/9283138722/
13/07/2013	14/07/2013	Montgomery	52.575052	-3.187258	https://www.flickr.com/photos/40011/9283136564/
13/07/2013	14/07/2013	Montgomery	52.575052	-3.187258	https://www.flickr.com/photos/40011/9280354905/
13/07/2013	14/07/2013	Montgomery	52.575052	-3.187258	https://www.flickr.com/photos/40011/9280352983/
13/07/2013	14/07/2013	Montgomery	52.575052	-3.187258	https://www.flickr.com/photos/40011/9280350645/
12/06/2021	12/06/2021	Allerdale District	54.921321	-3.162002	https://www.flickr.com/photos/linjohnpics/51247503561/

28/06/2016	28/06/2016	Taunton Deane Borough	51.025605	-3.077802	https://www.flickr.com/photos/pittypomm/27347786914/
30/05/2020	30/05/2020	Upper Cwmbran	51.669147	-3.064687	https://www.flickr.com/photos/16498755@N07/49951289828/
22/06/2014	22/06/2014	Taunton	51.023491	-3.062631	https://www.flickr.com/photos/john5554/14477592236/
22/06/2014	22/06/2014	Taunton	51.023497	-3.062628	https://www.flickr.com/photos/john5554/14314100788/
06/06/2016	17/06/2016	Bathpool	51.022952	-3.058837	https://www.flickr.com/photos/john5554/27661496731/
18/06/2015	25/06/2015	Bathpool	51.022811	-3.058831	https://www.flickr.com/photos/john5554/19115302986/
18/06/2015	25/06/2015	Bathpool	51.022972	-3.058756	https://www.flickr.com/photos/john5554/19141473995/
25/05/2017	04/06/2017	Bathpool	51.023047	-3.058673	https://www.flickr.com/photos/john5554/35094639015/
25/05/2017	04/06/2017	Bathpool	51.023052	-3.058634	https://www.flickr.com/photos/john5554/35094637925/
06/06/2016	17/06/2016	Bathpool	51.023130	-3.058612	https://www.flickr.com/photos/john5554/27661496931/
25/05/2017	04/06/2017	Bathpool	51.023033	-3.058545	https://www.flickr.com/photos/john5554/35094641225/
02/07/2015	03/07/2015	Bathpool	51.022686	-3.057664	https://www.flickr.com/photos/john5554/19204779759/
02/07/2015	03/07/2015	Bathpool	51.022652	-3.057623	https://www.flickr.com/photos/john5554/19203380780/
31/05/2020	31/05/2020	Cwmbran Central	51.652888	-3.026623	https://www.flickr.com/photos/16498755@N07/49956747937/
17/07/2021	17/07/2021	Panteg	51.674305	-3.025984	https://www.flickr.com/photos/16498755@N07/51316281677/
05/07/2016	10/07/2016	Monmouthshire	51.774598	-2.930409	https://www.flickr.com/photos/hoppy1951/27612624864/
10/06/2018	14/06/2018	Monmouthshire	51.774983	-2.930388	https://www.flickr.com/photos/hoppy1951/41892410865/

25/07/2016	22/08/2016	Monmouthshire	51.776822	-2.930012	https://www.flickr.com/photos/hoppy1951/29059903142/
21/07/2011	22/07/2011	Carlisle	54.903177	-2.918243	https://www.flickr.com/photos/8453647@N05/5963244189/
08/08/2015	08/08/2015	Ilton	50.966224	-2.915046	https://www.flickr.com/photos/iain_harris/20210136768/
15/06/2011	06/09/2011	Ruyton-XI-Towns	52.794666	-2.896745	https://www.flickr.com/photos/56765303@N04/6120241412/
31/05/2017	03/06/2017	Chester	53.190418	-2.880531	https://www.flickr.com/photos/99817330@N02/35066687905/
31/05/2017	21/06/2017	Chester	53.190350	-2.880488	https://www.flickr.com/photos/99817330@N02/35054376970/
05/08/2019	05/08/2019	Eccleston	53.154200	-2.875340	https://www.flickr.com/photos/131427325@N05/48467666802/
31/05/2021	31/05/2021	West Lancashire District	53.616854	-2.871776	https://www.flickr.com/photos/wistaston/51215575042/
26/06/2020	28/06/2020	West Lancashire District	53.620073	-2.865157	https://www.flickr.com/photos/66121367@N06/50054184337/
12/07/2020	12/07/2020	Stoke	53.254912	-2.861906	https://www.flickr.com/photos/p300njb/50104400178/
26/05/2012	27/05/2012	Clevedon	51.435614	-2.837176	https://www.flickr.com/photos/buttercup-pics/7278802728/
11/06/2017	24/06/2018	Thorney	50.995288	-2.811995	https://www.flickr.com/photos/rhubarbcrumbleandcustard/41170253480/
31/05/2015	02/06/2015	Thorney	50.994002	-2.809562	https://www.flickr.com/photos/rhubarbcrumbleandcustard/18191566950/
04/08/2012	09/09/2012	Rufford	53.632038	-2.806189	https://www.flickr.com/photos/callmeishmael2012/7964214328/
25/06/2017	25/06/2017	West Lancashire District	53.606811	-2.793053	https://www.flickr.com/photos/sleipnerofasgard/35492888606/
10/06/2015	15/06/2015	Somerset	51.024261	-2.791175	https://www.flickr.com/photos/rhubarbcrumbleandcustard/18840406825/
10/06/2015	15/06/2015	Somerset	51.020575	-2.780889	https://www.flickr.com/photos/rhubarbcrumbleandcustard/18654403029/

26/06/2017	27/06/2017	Somerset	51.153253	-2.779701	https://www.flickr.com/photos/bobs_retired_now/34766692163/
23/05/2015	25/05/2015	Lower Severalls	50.889555	-2.769678	https://www.flickr.com/photos/rhubarbcrumbleandcustard/17896439108/
18/06/2014	21/01/2015	Shrewsbury	52.700971	-2.733964	https://www.flickr.com/photos/dwilliams1971/16148318180/
02/06/2013	04/06/2013	Montacute	50.953636	-2.732551	https://www.flickr.com/photos/26368070@N08/8950428095/
13/08/2020	22/08/2020	Whitchurch Urban	52.955519	-2.718562	https://www.flickr.com/photos/47046427@N03/50254647802/
20/06/2019	08/04/2021	Ganarew	51.832652	-2.688099	https://www.flickr.com/photos/hoppy1951/51103668360/
01/06/2017	06/06/2017	Somerset	51.109584	-2.670321	https://www.flickr.com/photos/cosperwosper/34751058040/
18/05/2014	21/05/2014	Somerset	51.108901	-2.669366	https://www.flickr.com/photos/cosperwosper/14053841428/
18/05/2014	21/05/2014	Somerset	51.108901	-2.669366	https://www.flickr.com/photos/cosperwosper/14053841188/
02/07/2021	02/07/2021	Chorley District	53.709683	-2.656266	https://www.flickr.com/photos/189107290@N03/51286187329/
31/07/2019	09/08/2019	South Somerset District	51.006011	-2.645419	https://www.flickr.com/photos/29288836@N00/48498492222/
31/07/2019	03/08/2019	South Somerset District	51.006011	-2.645419	https://www.flickr.com/photos/29288836@N00/48446721402/
09/06/2011	09/06/2011	Bryn	53.503947	-2.629323	https://www.flickr.com/photos/misteroy/5815412061/
09/06/2011	09/06/2011	Bryn	53.503487	-2.627220	https://www.flickr.com/photos/misteroy/5815979744/
05/06/2015	08/06/2015	Clifton Maybank	50.922222	-2.599845	https://www.flickr.com/photos/rhubarbcrumbleandcustard/18409171169/
09/08/2011	11/08/2012	English Bicknor	51.860273	-2.590198	https://www.flickr.com/photos/gails_pictures/7759869242/
09/08/2011	11/08/2012	English Bicknor	51.861287	-2.590059	https://www.flickr.com/photos/gails_pictures/7759971464/
09/08/2011	11/08/2012	English Bicknor	51.861141	-2.589930	https://www.flickr.com/

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11/07/2013	11/07/2013	Bristol	51.454347	-2.538081	https://www.flickr.com/photos/bluesky4691/9262641253/
31/05/2017	01/06/2017	Wigan Metropolitan Borough	53.491897	-2.536731	https://www.flickr.com/photos/129605128@N04/34894188281/
31/05/2017	17/06/2017	Wigan Metropolitan Borough	53.491947	-2.536695	https://www.flickr.com/photos/129605128@N04/35358179355/
31/05/2017	25/08/2017	Wigan Metropolitan Borough	53.491888	-2.536587	https://www.flickr.com/photos/129605128@N04/36628596842/
26/06/2017	28/08/2017	Wigan Metropolitan Borough	53.491233	-2.536328	https://www.flickr.com/photos/129605128@N04/36054393043/
26/06/2017	27/06/2017	Wigan Metropolitan Borough	53.491188	-2.536281	https://www.flickr.com/photos/129605128@N04/34753691683/
26/06/2017	19/08/2017	Wigan Metropolitan Borough	53.491172	-2.536200	https://www.flickr.com/photos/129605128@N04/36527863481/
09/07/2018	13/07/2018	Cheshire	53.070434	-2.526700	https://www.flickr.com/photos/wright-leslie/43343111922/
10/06/2015	10/06/2015	Rudheath	53.249049	-2.494934	https://www.flickr.com/photos/98419471@N02/18495094788/
22/07/2012	05/06/2015	Rudheath	53.249049	-2.494934	https://www.flickr.com/photos/98419471@N02/18477709342/
26/05/2010	26/05/2010	Bristol	51.420687	-2.484605	https://www.flickr.com/photos/keehotee/4642270274/
25/06/2017	28/07/2017	Up Cerne	50.815992	-2.481923	https://www.flickr.com/photos/99817330@N02/36056672252/
25/06/2017	02/08/2017	Up Cerne	50.815999	-2.481880	https://www.flickr.com/photos/99817330@N02/35521797293/
23/07/2014	23/07/2014	Rixton	53.407625	-2.473576	https://www.flickr.com/photos/adymac/14541502337/
17/08/2020	18/08/2020	Weymouth and Portland District	50.618591	-2.463423	https://www.flickr.com/photos/peterharriman/50240151726/
21/07/2019	22/07/2019	Kelso	55.598593	-2.452021	https://www.flickr.com/photos/andymckie/48348471007/
29/07/2019	29/07/2019	Kelso	55.597904	-2.450852	https://www.flickr.com/photos/168615872@N02/

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21/07/2019	21/07/2019	Kelso	55.597904	-2.450852	https://www.flickr.com/photos/168615872@N02/48339861662/
06/06/2016	07/06/2016	Cadishead	53.417218	-2.448095	https://www.flickr.com/photos/adymac/26906749194/
13/05/2017	29/05/2017	Preens Eddy	52.616331	-2.441743	https://www.flickr.com/photos/dwilliams1971/34924216546/
01/06/2017	02/06/2017	Elton	53.131826	-2.406499	https://www.flickr.com/photos/wistaston/34237399593/
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01/06/2017	02/06/2017	Elton	53.131826	-2.406499	https://www.flickr.com/photos/wistaston/34237399193/
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05/07/2019	05/07/2019	Moston	53.136557	-2.401585	https://www.flickr.com/photos/wistaston/48207348232/
14/06/2014	14/06/2014	Altrincham	53.380191	-2.399634	https://www.flickr.com/photos/31693460@N06/14417323831/
06/06/2018	19/06/2018	Trafford Metropolitan Borough	53.380319	-2.399525	https://www.flickr.com/photos/31693460@N06/41091444940/
03/07/2012	04/07/2012	Altrincham	53.380050	-2.399492	https://www.flickr.com/photos/31693460@N06/7501003878/
29/05/2016	31/05/2016	Trafford Metropolitan Borough	53.380383	-2.399403	https://www.flickr.com/photos/31693460@N06/27281288642/
19/05/2014	22/05/2014	Altrincham	53.379925	-2.399225	https://www.flickr.com/photos/31693460@N06/14237613292/
19/05/2014	20/05/2014	Altrincham	53.379925	-2.399225	https://www.flickr.com/photos/31693460@N06/14036580019/
19/05/2014	19/05/2014	Altrincham	53.379925	-2.399225	https://www.flickr.com/photos/31693460@N06/14036630960/
09/06/2017	23/06/2017	Trafford Metropolitan Borough	53.379905	-2.399209	https://www.flickr.com/photos/31693460@N06/35422069346/
05/06/2015	06/06/2015	Trafford Metropolitan Borough	53.379883	-2.399070	https://www.flickr.com/photos/31693460@N06/17909994874/

16/06/2013	09/07/2013	Altrincham	53.380205	-2.398967	https://www.flickr.com/photos/31693460@N06/9184270647/
12/06/2021	12/06/2021	Sandbach	53.131729	-2.394536	https://www.flickr.com/photos/wistaston/51242663526/
12/06/2021	12/06/2021	Sandbach	53.131729	-2.394536	https://www.flickr.com/photos/wistaston/51241954137/
12/06/2021	12/06/2021	Sandbach	53.131729	-2.394536	https://www.flickr.com/photos/wistaston/51243720180/
12/06/2021	12/06/2021	Sandbach	53.131729	-2.394536	https://www.flickr.com/photos/wistaston/51242663291/
20/06/2015	20/06/2015	Combe Hay	51.335263	-2.391221	https://www.flickr.com/photos/61830788@N03/18800235678/
04/08/2019	05/08/2019	Sandbach	53.131150	-2.389751	https://www.flickr.com/photos/wistaston/48461180697/
19/06/2013	23/06/2013	Wheelock	53.128704	-2.377939	https://www.flickr.com/photos/wistaston/9113824331/
28/05/2016	08/06/2016	Hampton	52.481708	-2.375128	https://www.flickr.com/photos/dwilliams1971/27546706255/
06/08/2019	06/08/2019	Sandbach	53.129894	-2.366040	https://www.flickr.com/photos/wistaston/48475437511/
05/08/2019	05/08/2019	Sandbach	53.129875	-2.365965	https://www.flickr.com/photos/wistaston/48465231292/
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25/07/2021	25/07/2021	Hassall	53.125143	-2.343006	https://www.flickr.com/photos/wistaston/51333743628/
25/07/2021	25/07/2021	Hassall	53.125143	-2.343006	https://www.flickr.com/photos/wistaston/51333529561/
02/06/2016	03/06/2016	Bath	51.396139	-2.339680	https://www.flickr.com/photos/ianredding/27334171112/
30/05/2016	31/05/2016	Walcot	51.395939	-2.338864	https://www.flickr.com/photos/ianredding/27092397790/
09/06/2016	11/06/2016	Bath	51.397579	-2.337384	https://www.flickr.com/photos/ianredding/27596240845/

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25/06/2021	21/07/2021	Tewkesbury	51.979364	-2.149546	https://www.flickr.com/photos/129733140@N05/51326712453/
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01/06/2014	01/06/2014	Stockport	53.439068	-2.141647	https://www.flickr.com/photos/jvarley_19/14133499899/
13/07/2013	15/07/2013	Stafford	52.808144	-2.139308	https://www.flickr.com/photos/84646892@N03/9297057308/
13/07/2013	15/07/2013	Stafford	52.807703	-2.138235	https://www.flickr.com/photos/84646892@N03/9294276701/
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31/07/2020	23/08/2020	Wychavon District	52.229061	-2.110184	https://www.flickr.com/photos/jrc1/50258035143/
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29/05/2015	07/06/2015	Ratford	51.441041	-2.029848	https://www.flickr.com/photos/oldforgecam/18538212796/
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23/07/2021	25/07/2021	Bird End	52.525096	-1.963677	https://www.flickr.com/photos/michaeljh/51334590659/
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21/07/2017	24/07/2017	Halifax	53.698372	-1.877048	https://www.flickr.com/photos/ukrabbiter/35731102390/
25/07/2014	25/07/2014	Wixford	52.189062	-1.873930	https://www.flickr.com/photos/kimbenson45/14720983156/
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26/04/2020	27/04/2020	Iford	50.743557	-1.814718	https://www.flickr.com/photos/154348953@N08/49824748681/
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31/05/2017	05/06/2017	Ibsley	50.885772	-1.793581	https://www.flickr.com/photos/colinlamond/34310911133/
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02/08/2019	06/08/2019	Witton-le-Wear	54.675034	-1.749583	https://www.flickr.com/photos/gilli8888/48474917261/
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08/07/2018	08/07/2018	Stratford-upon-Avon	52.183461	-1.707998	https://www.flickr.com/photos/141425391@N07/43235280712/
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25/05/2019	27/05/2019	Tamworth	52.627411	-1.696723	https://www.flickr.com/photos/richardjameswhite/47943969202/
25/05/2019	27/05/2019	Tamworth	52.627411	-1.696723	https://www.flickr.com/photos/richardjameswhite/47943980073/
25/05/2019	27/05/2019	Fazeley	52.621519	-1.694434	https://www.flickr.com/photos/richardjameswhite/47943971667/
06/06/2018	15/06/2018	Bridgetown	52.195356	-1.694279	https://www.flickr.com/photos/davea2007/42089567674/
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21/05/2015	21/05/2015	Cole End	52.522168	-1.688525	https://www.flickr.com/photos/31339850@N06/17932970621/
21/05/2015	21/05/2015	Lea Marston	52.522325	-1.687474	https://www.flickr.com/photos/31339850@N06/17932981201/
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29/05/2018	29/05/2018	Marston	52.548351	-1.687281	https://www.flickr.com/photos/92610731@N05/42387192552/
29/05/2018	29/05/2018	Marston	52.548351	-1.687281	https://www.flickr.com/photos/92610731@N05/42387192232/
30/05/2012	30/05/2012	Lea Marston	52.524159	-1.687088	https://www.flickr.com/

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02/07/2014	02/07/2014	Lea Marston	52.524219	-1.680767	https://www.flickr.com/photos /31339850@N06/ 14371391018/
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19/05/2010	19/05/2010	Coleshill	51.637187	-1.661794	https://www.flickr.com/ photos/od0man/4622803486/
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03/08/2012	07/08/2012	King's Lynn	52.764397	0.441448	https://www.flickr.com/photos/-peregrine-falcon-/7732652138/
04/08/2019	07/08/2019	King's Lynn and West Norfolk District	52.868433	0.451500	https://www.flickr.com/photos/bramblejungle/48482056977/
27/05/2016	27/05/2016	Chelmsford	51.732777	0.460348	https://www.flickr.com/photos/rivercrouchwalker/27190445962/
04/07/2013	07/07/2013	Chelmsford	51.743997	0.475115	https://www.flickr.com/photos/byneilhall/9231754171/
12/07/2014	13/07/2014	Chelmsford	51.728603	0.485651	https://www.flickr.com/photos/bruce82/14455704148/
21/07/2013	23/07/2013	Chelmsford	51.729845	0.497002	https://www.flickr.com/photos/rivercrouchwalker/

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15/06/2014	15/06/2014	Hockwold cum Wilton	52.446639	0.500414	https://www.flickr.com/photos/aljones27/14405606016/
05/07/2016	11/07/2016	Forest Heath District	52.448852	0.519611	https://www.flickr.com/photos/xaotearoax/28156707611/
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10/06/2012	12/06/2012	Lakenheath	52.447143	0.526678	https://www.flickr.com/photos/chescott/7365087830/
24/05/2012	25/05/2012	Hockwold cum Wilton	52.450543	0.528448	https://www.flickr.com/photos/dave27/7269026554/
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23/06/2011	24/06/2011	Lakenheath	52.448181	0.536580	https://www.flickr.com/photos/matpreec77/5865419157/
22/05/2017	23/05/2017	Essex	51.748955	0.561780	https://www.flickr.com/photos/hc1/34683158992/
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15/01/2011	15/01/2011	Icklingham	52.320441	0.602703	https://www.flickr.com/photos/mark_at_magdalen/5357828013/
02/06/2011	15/02/2013	Lackford	52.304253	0.639266	https://www.flickr.com/photos/john_bugg/8475894254/
28/05/2020	29/05/2020	Maldon District	51.774340	0.642281	https://www.flickr.com/photos/stuart166axe/49947329791/
21/06/2011	21/06/2011	Braintree District	51.947699	0.642818	https://www.flickr.com/photos/kitmasterbloke/5856712897/
28/05/2020	09/07/2020	St. Edmundsbury District	52.305127	0.648171	https://www.flickr.com/photos/bobchappell/50092614998/
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24/06/2019	07/07/2019	St. Edmundsbury District	52.305036	0.648855	https://www.flickr.com/photos/bobchappell/48218902076/
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21/06/2019	25/08/2019	St. Edmundsbury District	52.305038	0.648863	https://www.flickr.com/photos/bobchappell/48615731541/
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15/06/2020	02/08/2020	St. Edmundsbury District	52.305055	0.648947	https://www.flickr.com/photos/bobchappell/50178707333/
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28/05/2020	22/12/2020	St. Edmundsbury District	52.305661	0.649266	https://www.flickr.com/photos/bobchappell/50746533651/
10/05/2017	10/05/2017	Little Braxted	51.791944	0.651458	https://www.flickr.com/photos/hc1/34447774941/
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13/07/2018	17/07/2018	Essex	51.578829	0.663385	https://www.flickr.com/photos/51885768@N03/42571195375/
08/06/2013	08/06/2013	Glemsford	52.088366	0.684263	https://www.flickr.com/photos/9701017@N08/8989208651/
03/06/2011	27/12/2013	Santon Downham	52.452021	0.689864	https://www.flickr.com/photos/dave_p_brecks/11590837793/
14/07/2015	15/07/2015		52.452613	0.690577	https://www.flickr.com/photos/92329438@N05/19711824192/
26/06/2012	03/07/2012	Liston	52.073727	0.707050	https://www.flickr.com/photos/cobaltfish/7497352068/
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29/07/2019	29/07/2019	Colchester District	51.896755	0.882725	https://www.flickr.com/photos/postmanpetecoluk/48406656147/
22/05/2011	23/05/2011	Colchester	51.846588	0.885123	https://www.flickr.com/photos/petrochemist/5752893194/
22/05/2011	23/05/2011	Colchester	51.846492	0.885509	https://www.flickr.com/photos/petrochemist/5752892958/
06/08/2016	10/08/2016	Norfolk	52.930417	0.890858	https://www.flickr.com/photos/65272192@N02/28896720475/
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08/08/2019	08/08/2019	Ashford	51.224236	0.969083	https://www.flickr.com/photos/smudge9000/48489553076/
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29/05/2020	29/05/2020	Mid Suffolk District	52.161103	1.046965	https://www.flickr.com/photos/caught-on-digital/49949399593/
29/05/2020	29/05/2020	Mid Suffolk District	52.161103	1.046965	https://www.flickr.com/photos/caught-on-digital/49949899761/
29/05/2020	29/05/2020	Mid Suffolk District	52.163274	1.048199	https://www.flickr.com/photos/caught-on-digital/49949397768/
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08/06/2016	10/06/2016	Grove Ferry	51.317349	1.194934	https://www.flickr.com/photos/markkilner/27548986746/
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10/06/2016	10/06/2016	Colney	52.626993	1.215212	https://www.flickr.com/photos/phil-carpenter/27476076362/
28/06/2012	28/06/2012	Norwich	52.617693	1.244716	https://www.flickr.com/photos/trevsphotographs/7462006800/
26/07/2013	26/07/2013	Norwich	52.641566	1.283941	https://www.flickr.com/photos/aaronmstanley/9368651341/
23/06/2014	23/06/2014	Norwich	52.636631	1.287846	https://www.flickr.com/photos/69770179@N06/14491011422/
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12/05/2011	13/05/2011	Norwich	52.612507	1.313080	https://www.flickr.com/photos/f3liney/5714312247/
11/06/2014	19/06/2014	Minster	51.318048	1.313630	https://www.flickr.com/photos/markkilner/14271680888/
11/06/2014	11/06/2014	Minster	51.318048	1.313630	https://www.flickr.com/photos/markkilner/14398364162/
01/07/2019	14/07/2019	Thanet District	51.318181	1.313703	https://www.flickr.com/photos/markkilner/48284176517/
01/07/2019	04/07/2019		51.318188	1.313703	https://www.flickr.com/photos/markkilner/48198465682/
28/07/2012	31/07/2012	Wroxham	52.702966	1.384062	https://www.flickr.com/photos/39942205@N03/7683916994/
04/07/2016	08/06/2017	Flixton	52.435956	1.396669	https://www.flickr.com/photos/amks_photos/35010829722/
15/06/2016	15/06/2016	Surlingham	52.599137	1.431698	https://www.flickr.com/photos/fridda_1/27696127135/
14/05/2014	15/05/2014	Surlingham	52.599444	1.433833	https://www.flickr.com/photos/65272192@N02/

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14/05/2014	15/05/2014	Surlingham	52.599444	1.433833	https://www.flickr.com/photos/65272192@N02/14191128772/
28/05/2020	28/05/2020	South Norfolk District	52.605266	1.437482	https://www.flickr.com/photos/nikonjeremy/49945734648/
28/05/2020	28/05/2020	South Norfolk District	52.605266	1.437482	https://www.flickr.com/photos/nikonjeremy/49946527337/
28/05/2020	28/05/2020	South Norfolk District	52.605266	1.437482	https://www.flickr.com/photos/nikonjeremy/49946527567/
25/05/2014	25/05/2014	Brundall	52.609596	1.439819	https://www.flickr.com/photos/76985033@N02/14287408443/
25/05/2014	25/05/2014	Surlingham	52.607884	1.440462	https://www.flickr.com/photos/76985033@N02/14080611798/
11/06/2018	10/07/2018	Broadland District	52.604558	1.444227	https://www.flickr.com/photos/oandrews/41516877660/
29/05/2017	29/05/2017	Broadland District	52.603282	1.448307	https://www.flickr.com/photos/nrgoodrum/34933358226/
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09/07/2013	13/07/2013	Buckenham	52.603113	1.448830	https://www.flickr.com/photos/jdoakey/9274356879/
10/06/2014	10/06/2014	Strumpshaw	52.607517	1.455860	https://www.flickr.com/photos/hatto26/14204148038/
25/05/2014	25/05/2014	Strumpshaw	52.607517	1.455860	https://www.flickr.com/photos/hatto26/14266046224/
01/06/2014	10/06/2014	Strumpshaw	52.606527	1.456466	https://www.flickr.com/photos/jdoakey/14203126750/
06/07/2015	13/07/2016	Gillingham	52.467338	1.533901	https://www.flickr.com/photos/nikonjeremy/27998668880/
24/07/2020	07/08/2020	Suffolk Coastal District	52.240891	1.591277	https://www.flickr.com/photos/soresha/50199641803/
24/07/2020	07/08/2020	Suffolk Coastal District	52.240880	1.591316	https://www.flickr.com/photos/soresha/50199640848/
24/07/2020	07/08/2020	Suffolk Coastal District	52.240880	1.591350	https://www.flickr.com/photos/soresha/

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24/07/2020	07/08/2020	Suffolk Coastal District	52.240875	1.591355	https://www.flickr.com/photos/soresha/50200452567/
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13/06/2021	07/07/2021	Mountain Ash	51.688088	-3.394153	https://www.flickr.com/photos/129802601@N03/51296742776/
15/08/2021	15/08/2021	Rother District	50.869213	0.502442	https://www.flickr.com/photos/nilocspics/51380350940/
15/05/2020	15/05/2020	Guildford District	51.315458	-0.454752	https://www.flickr.com/photos/14902568@N05/49898768541/
10/06/2016	10/06/2016	London	51.563945	-0.039782	https://www.flickr.com/photos/hawkeye2011/27551517506/
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14/06/2021	26/06/2021	Worcester District	52.183296	-2.222833	https://www.flickr.com/photos/davea2007/51271369198/
14/06/2021	26/06/2021	Worcester District	52.183296	-2.222833	https://www.flickr.com/photos/davea2007/51270443397/
14/06/2021	26/06/2021	Worcester District	52.183298	-2.222832	https://www.flickr.com/photos/davea2007/51272218780/
14/06/2021	26/06/2021	Worcester District	52.183298	-2.222832	https://www.flickr.com/photos/davea2007/51272218545/
14/06/2021	26/06/2021	Worcester District	52.183299	-2.222832	https://www.flickr.com/photos/davea2007/51271921344/
26/07/2019	15/08/2019	Bradmore	52.585298	-2.176866	https://www.flickr.com/photos/davea2007/48540811456/
02/07/2018	07/07/2018	Wolverhampton Metropolitan Borough	52.606756	-2.145829	https://www.flickr.com/photos/davea2007/29377918698/
25/05/2019	27/05/2019	Tamworth	52.627730	-1.696817	https://www.flickr.com/photos/richardjameswhite/47943983338/
09/06/2013	11/06/2013	Dunbridge	51.042888	-1.531567	https://www.flickr.com/photos/stevejm2009/

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25/05/2017	08/02/2021	Taunton Deane District	51.023052	-3.058634	https://www.flickr.com/photos/john5554/50922241983/
17/06/2017	26/01/2022	Taunton Deane District	51.022986	-3.058509	https://www.flickr.com/photos/john5554/51845706655/
23/05/2014	Not Provided	West Bridgford	52.942741	-1.129446	https://www.flickr.com/photos/42985099@N03/14066322059
06/06/2016	17/06/2016	Bathpool	51.023133	-3.059253	https://www.flickr.com/photos/john5554/27661496831/
25/05/2016	27/05/2016	Bathpool	51.023108	-3.058681	https://www.flickr.com/photos/john5554/27293850275/
25/05/2016	27/05/2016	Bathpool	51.023088	-3.058662	https://www.flickr.com/photos/john5554/27293849995/
17/06/2017	30/06/2017	Bathpool	51.022986	-3.058509	https://www.flickr.com/photos/john5554/35238471430/
17/06/2017	30/06/2017	Bathpool	51.022927	-3.058503	https://www.flickr.com/photos/john5554/34783971344/
25/05/2017	04/06/2017	Bathpool	51.022880	-3.058378	https://www.flickr.com/photos/john5554/35094634025/
12/06/2018	17/06/2018	Bathpool	51.022488	-3.057003	https://www.flickr.com/photos/john5554/42144871764/
17/07/2021	17/07/2021	Panteg	51.674305	-3.025984	https://www.flickr.com/photos/16498755@N07/51317742764/
24/05/2016	24/05/2016	Somerset	51.108901	-2.669366	https://www.flickr.com/photos/cosperwosper/27222481295/
09/07/2018	01/08/2018	Cheshire	53.070434	-2.526700	https://www.flickr.com/photos/wright-leslie/41971428790/
09/06/2016	11/06/2016	Bath	51.397579	-2.337384	https://www.flickr.com/photos/ianredding/26987459223/
17/10/2012	17/10/2012	Briantspuddle	50.737840	-2.261252	https://www.flickr.com/photos/drinkermoth/8097509402/
23/05/2017	26/05/2017	South Yorkshire	53.423524	-1.368688	https://www.flickr.com/photos/peter_quinn1/

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07/06/2018	13/06/2018	Rotherham Metropolitan Borough	53.423849	-1.367197	https://www.flickr.com/photos/peter_quinn1/40968411040/
09/07/2011	09/07/2011	Oxford	51.764077	-1.249084	https://www.flickr.com/photos/jacaroo/5919907108/
09/07/2011	09/07/2011	Oxford	51.764127	-1.249050	https://www.flickr.com/photos/jacaroo/5919904922/
24/05/2014	25/05/2014	West Bridgford	52.942741	-1.129446	https://www.flickr.com/photos/odonataman/14074190028/
21/01/2015	21/01/2015	Nottingham	52.942891	-1.128973	https://www.flickr.com/photos/odonataman/16308968686/
13/01/2015	13/01/2015	Nottingham	52.942891	-1.128973	https://www.flickr.com/photos/odonataman/16085965178/
22/12/2014	24/12/2014	Nottingham	52.942891	-1.128973	https://www.flickr.com/photos/odonataman/16070808036/
21/12/2014	21/12/2014	Nottingham	52.942891	-1.128973	https://www.flickr.com/photos/odonataman/16073608875/
04/10/2014	05/10/2014	Nottingham	52.942891	-1.128973	https://www.flickr.com/photos/odonataman/15439827655/
24/06/2014	26/06/2014	Nottingham	52.942891	-1.128973	https://www.flickr.com/photos/odonataman/14507063445/
26/05/2014	27/05/2014	West Bridgford	52.942935	-1.126484	https://www.flickr.com/photos/odonataman/14252364726/
29/05/2014	02/06/2014	West Bridgford	52.942948	-1.125884	https://www.flickr.com/photos/odonataman/14297659121/
16/05/2014	19/05/2014	Cosgrove	52.072120	-0.840797	https://www.flickr.com/photos/watterbug/14220460225/
08/06/2020	19/06/2020	Leighton-Linslade	51.928900	-0.671806	https://www.flickr.com/photos/steveinleightonsphotos/50022796821/
19/06/2016	25/06/2016	Aldwincle	52.414919	-0.515195	https://www.flickr.com/photos/oandrews/27817756651/
08/06/2015	11/06/2015	Aldwincle	52.415594	-0.514437	https://www.flickr.com/

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28/05/2015	08/06/2015	London	51.402043	-0.340147	https://www.flickr.com/ photos/sjdarlington/ 17988752213/
03/07/2012	28/11/2013	Huntingdon	52.321335	-0.185437	https://www.flickr.com/ photos/ leohillierphotography/ 11104336273/
30/06/2015	13/08/2015	Litcham	52.719320	0.789706	https://www.flickr.com/ photos/16613507@N02/ 20556544391/
13/06/2014	13/06/2014	Canterbury	51.273632	1.062521	https://www.flickr.com/ photos/85916010@N03/ 14227806449/
10/06/2012	10/06/2012	Ipswich	52.029874	1.132310	https://www.flickr.com/ photos/deviantlight/ 7359031322/
30/06/2021	18/01/2022	South Derbyshire District	52.882611	-1.335085	https://www.flickr.com/ photos/pjfulford/ 51829193798/
22/06/2021	23/06/2021	Fulford	53.928280	-1.082485	https://www.flickr.com/ photos/smithandjones/ 51266154754/
22/07/2021	30/07/2021	Clyne and Melincourt	51.703080	-3.722503	https://www.flickr.com/ photos/matthew_witty/ 51346929879/
12/06/2018	24/06/2018	Bathpool	51.022430	-3.056998	https://www.flickr.com/ photos/john5554/ 42270342424/
01/06/2021	02/06/2021	West Lancashire District	53.616789	-2.871830	https://www.flickr.com/ photos/31693460@N06/ 51221567934/
10/06/2015	15/06/2015	Somerset	51.024258	-2.791037	https://www.flickr.com/ photos/rhubarbcrumbleand custard/18652424750/
21/07/2019	07/11/2019	Mendip District	51.191744	-2.783995	https://www.flickr.com/ photos/colinlamond/ 49030469737/
12/07/2017	12/07/2017	Osbaston	51.818525	-2.714837	https://www.flickr.com/ photos/16498755@N07/ 35070553393/
18/05/2011	19/05/2011	Cound Moor	52.629989	-2.653369	https://www.flickr.com/ photos/johnbalcombe/ 5736339370/
31/05/2015	31/05/2015	Shropshire	52.625401	-2.466795	https://www.flickr.com/ photos/eddie_austrums/ 18288214096/
16/05/2019	17/05/2019	Elton	53.131826	-2.406499	https://www.flickr.com/ photos/wistaston/ 47867488011/

06/07/2013	07/07/2013	Upper Arley	52.417736	-2.349612	https://www.flickr.com/photos/31339850@N06/9232015754/
24/08/2019	24/08/2019	Bradmore	52.586611	-2.174041	https://www.flickr.com/photos/davea2007/48612350298/
30/05/2020	06/06/2020	Cherwell District	52.039960	-1.299734	https://www.flickr.com/photos/ad2812/49976133843/
22/05/2012	22/05/2012	Haversham	52.072608	-0.810263	https://www.flickr.com/photos/fatphotographer/7249361054/
21/07/2013	22/07/2013	Elstead	51.186855	-0.722415	https://www.flickr.com/photos/cliffbuckton/9340182047/
31/05/2017	31/05/2017	Thurlby	53.134034	-0.642099	https://www.flickr.com/photos/lincsbirder/34628389950/
20/05/2017	20/05/2017	Thurlby	53.134034	-0.642099	https://www.flickr.com/photos/lincsbirder/33934648504/
25/05/2014	25/05/2014	Surlingham	52.607070	1.441440	https://www.flickr.com/photos/76985033@N02/14263876811/
20/06/2018	22/07/2018	Eastbridge	52.241128	1.591440	https://www.flickr.com/photos/99817330@N02/41754712570/
02/06/2021	04/06/2021	Waveney District	52.462324	1.633465	https://www.flickr.com/photos/melopics/51224950875/

Appendix B: Odonata and Hymenoptera distribution and niche change

Study species within the Odonata and Hymenoptera groups were selected as those which have increased in distribution size in Great Britain during a recent period of climate change whilst remaining within a consistent climate niche. Species' occupancy was determined by comparing two periods, T1 (1961-1980) and T2 (2001-2020), using the Telfer index. Species' realised climate niche was also compared between T1 and T2 including Schener's D index of niche overlap, expansion (climate niche occupied at T2 only), and contraction (climate niche occupied at T1 only). A niche similarity test was also computed to determine whether species' climate niche was statistically similar at T2 compared to T1. The selected Odonatan (table B.1) and Hymenopteran (table B.2) study species are indicated in blue – those species with a positive increase (Telfer Index > 0) which exhibited niche conservatism (Conservatism p < 0.05). Tables are ordered from species with the greatest increase in distribution size to the greatest decrease.

Table B.1. Distribution and climate niche change for Odonatan species in Great Britain between 1961-1980 and 2001-2020 including occupancy change, niche overlap, niche expansion, niche contraction, and the p-value for a statistical test for niche conservatism between periods. Species indicated in blue are those with a positive increase in distribution (Telfer Index > 0) which exhibited niche conservatism (p < 0.05).

Species	Telfer Index	Overlap	Expansion	Contraction	Conservatism p
Sympetrum sanguineum	1.91	0.24	0.41	0.18	0.089
Aeshna mixta	1.74	0.24	0.29	0.17	0.030
Libellula fulva	1.69	0.03	0.87	0.57	0.109
Anax imperator	1.68	0.35	0.09	0.10	0.010
Lestes dryas	1.50	0.00	0.99	0.97	0.317
Orthetrum cancellatum	1.30	0.26	0.27	0.18	0.040
Anaciaeschna isoceles	1.23	0.00	1.00	1.00	1.000
Brachytron pratense	0.96	0.12	0.56	0.19	0.139
Erythromma najas	0.72	0.11	0.74	0.38	0.139
Aeshna cyanea	0.65	0.34	0.20	0.13	0.030
Sympetrum striolatum	0.61	0.37	0.04	0.06	0.010
Libellula depressa	0.57	0.41	0.07	0.09	0.030
Calopteryx splendens	0.48	0.34	0.36	0.18	0.040
Somatochlora arctica	0.44	0.21	0.32	0.04	0.010
Libellula quadrimaculata	0.42	0.23	0.05	0.01	0.059
Sympetrum flaveolum	0.23	0.08	0.82	0.49	0.168
Platycnemis pennipes	0.23	0.29	0.55	0.24	0.050
Pyrrhosoma nymphula	-0.01	0.31	0.04	0.01	0.040
Coenagrion puella	-0.06	0.42	0.12	0.07	0.010
Calopteryx virgo	-0.20	0.33	0.11	0.05	0.030
Enallagma cyathigerum	-0.30	0.39	0.04	0.03	0.059
Aeshna caerulea	-0.30	0.36	0.20	0.15	0.010
Coenagrion hastulatum	-0.45	0.09	0.90	0.66	0.149
Gomphus vulgatissimus	-0.56	0.13	0.76	0.65	0.059
Coenagrion mercuriale	-0.63	0.06	0.61	0.40	0.208

Cordulia aenea	-0.63	0.05	0.67	0.36	0.287
Lestes sponsa	-0.64	0.29	0.05	0.02	0.099
Aeshna grandis	-0.65	0.31	0.36	0.31	0.059
Orthetrum coerulescens	-0.68	0.25	0.12	0.10	0.079
Aeshna juncea	-0.76	0.24	0.02	0.01	0.030
Cordulegaster boltonii	-0.80	0.25	0.05	0.01	0.040
Ischnura elegans	-1.00	0.40	0.07	0.06	0.030
Ischnura pumilio	-1.12	0.34	0.17	0.09	0.030
Somatochlora metallica	-1.16	0.05	0.84	0.03	0.139
Sympetrum danae	-1.29	0.27	0.03	0.02	0.069
Ceriagrion tenellum	-1.34	0.33	0.10	0.28	0.040
Coenagrion pulchellum	-1.34	0.15	0.41	0.32	0.119
Leucorrhinia dubia	-1.96	0.13	0.28	0.01	0.030

Table B.2. Distribution and climate niche change for Hymenopteran species in Great Britain between 1961-1980 and 2001-2020 including occupancy change, niche overlap, niche expansion, niche contraction, and the p-value for a statistical test for niche conservatism between periods. Species indicated in blue are those with a positive increase in distribution (Telfer Index > 0) which exhibited niche conservatism (p < 0.05).

Species	Telfer Index	Overlap	Expansion	Contraction	Conservatism p
Andrena cineraria	2.67	0.24	0.10	0.28	0.020
Apis mellifera	2.64	0.18	0.25	0.10	0.119
Bombus lucorum agg	2.63	0.13	0.33	0.01	0.079
Nomada goodeniana	2.63	0.15	0.60	0.22	0.069
Anthophora plumipes	2.45	0.08	0.29	0.22	0.069
Andrena nitida	2.32	0.04	0.79	0.22	0.168
Cerceris rybyensis	2.28	0.05	0.70	0.28	0.119
Megachile ligniseca	2.20	0.00	1.00	0.97	0.356
Andricus quercuscalicis	1.93	0.16	0.84	0.04	0.010
Anthidium manicatum	1.91	0.07	0.58	0.06	0.129
Tiphia femorata	1.89	0.06	0.52	0.14	0.198
Lasioglossum lativentre	1.86	0.16	0.60	0.20	0.129
Lasius niger agg	1.76	0.08	0.47	0.36	0.168
Anoplius nigerrimus	1.69	0.09	0.33	0.07	0.109
Nomada fabriciana	1.69	0.10	0.59	0.19	0.079
Cerceris arenaria	1.64	0.15	0.44	0.32	0.030
Hylaeus signatus	1.61	0.02	0.86	0.13	0.129
Trichrysis cyanea	1.50	0.11	0.72	0.37	0.129
Megachile willughbiella	1.50	0.14	0.41	0.14	0.089
Ectemnius lituratus	1.48	0.05	0.77	0.16	0.198
Osmia leaiana	1.44	0.09	0.49	0.39	0.099
Mimesa lutaria	1.44	0.05	0.81	0.48	0.119
Andrena dorsata	1.40	0.14	0.32	0.13	0.158
Myrmica scabrinodis agg	1.36	0.04	0.34	0.05	0.248
Pepsis cyanea	1.34	0.06	0.94	0.70	0.129
Osmia bicolor	1.31	0.01	0.96	0.70	0.277

Symmorphus gracilis	1.30	0.17	0.63	0.41	0.109
Bombus jonellus	1.29	0.06	0.33	0.00	0.366
Ectemnius continuus	1.29	0.12	0.30	0.24	0.158
Anoplius infuscatus	1.28	0.10	0.37	0.27	0.109
Nomada leucophthalma	1.27	0.22	0.46	0.13	0.059
Heriades truncorum	1.27	0.00	1.00	1.00	1.000
Phymatocera aterrima	1.27	0.09	0.48	0.15	0.218
Andrena bimaculata	1.23	0.01	0.97	0.60	0.297
Lasius brunneus	1.22	0.02	0.95	0.76	0.198
Profenusa pygmaea	1.22	0.02	1.00	0.66	0.178
Hedychridium roseum	1.22	0.02	0.90	0.71	0.208
Melitta haemorrhoidalis	1.21	0.01	0.90	0.37	0.257
Ancistrocerus parietum	1.20	0.05	0.82	0.33	0.139
Odynerus spinipes	1.20	0.17	0.54	0.16	0.079
Andrena praecox	1.20	0.08	0.83	0.31	0.109
Pseudomalus auratus	1.20	0.05	0.62	0.66	0.287
Priocnemis perturbator	1.19	0.12	0.60	0.10	0.109
Symmorphus bifasciatus	1.19	0.03	0.93	0.60	0.297
Andrena minutuloides	1.19	0.04	0.82	0.59	0.139
Osmia spinulosa	1.18	0.09	0.56	0.47	0.178
Hedychridium ardens	1.17	0.02	0.79	0.29	0.218
Amblyteles armatorius	1.16	0.08	0.92	0.39	0.139
Dipogon subintermedius	1.14	0.02	0.98	0.59	0.109
Lasioglossum prasinum	1.13	0.04	0.76	0.57	0.198
Andrena wilkella	1.12	0.12	0.56	0.13	0.119
Lasioglossum malachurum	1.11	0.03	0.85	0.00	0.188
Bombus rupestris	1.11	0.16	0.20	0.21	0.218
Auplopus carbonarius	1.10	0.03	0.71	0.50	0.218
Chelostoma florisomne	1.09	0.18	0.64	0.36	0.109
Osmia bicornis	1.09	0.16	0.44	0.17	0.099
Hylaeus dilatatus	1.08	0.02	0.81	0.04	0.158
Astata boops	1.07	0.01	0.93	0.58	0.168
Lasioglossum albipes	1.07	0.10	0.19	0.06	0.099
Nomada rufipes	1.06	0.08	0.57	0.19	0.109
Chrysis angustula	1.05	0.12	0.46	0.40	0.099
Andrena scotica	1.03	0.08	0.56	0.07	0.149
Nysson spinosus	1.03	0.12	0.55	0.07	0.168
Sphecodes geoffrellus	1.03	0.09	0.42	0.09	0.099
Bombus soroeensis	1.02	0.07	0.71	0.00	0.020
Episyron rufipes	1.01	0.03	0.52	0.19	0.218
Eumenes coarctatus	1.01	0.00	1.00	1.00	0.228
Pontania pedunculi	0.99	0.05	0.73	0.00	0.129
Arge cyanocrocea	0.99	0.15	0.46	0.27	0.099
Anthophora bimaculata	0.99	0.02	0.72	0.42	0.277
Halictus rubicundus	0.97	0.11	0.29	0.08	0.218
Diplolepis rosae	0.96	0.25	0.57	0.13	0.129

Cynips quercusfolii	0.96	0.24	0.15	0.13	0.069
Andrena subopaca	0.96	0.15	0.46	0.04	0.099
Lindenius panzeri	0.96	0.00	1.00	0.63	0.198
Ancistrocerus gazella	0.95	0.02	0.55	0.31	0.257
Lasioglossum calceatum	0.95	0.14	0.17	0.03	0.139
Sphecodes ephippius	0.95	0.14	0.40	0.13	0.139
Chrysis impressa	0.95	0.02	0.83	0.01	0.129
Andrena trimmerana	0.94	0.02	0.61	0.36	0.277
Andrena nigroaenea	0.93	0.10	0.50	0.21	0.079
Bombus magnus	0.93	0.04	0.67	0.00	0.455
Sphecodes reticulatus	0.93	0.00	0.99	0.28	0.139
Agenioideus cinctellus	0.92	0.15	0.40	0.29	0.069
Lasioglossum leucozonium	0.90	0.02	0.66	0.09	0.248
Nomada panzeri	0.90	0.24	0.35	0.09	0.040
Trypoxylon figulus	0.90	0.13	0.87	0.51	0.089
Andrena haemorrhoa	0.89	0.16	0.49	0.08	0.069
Priocnemis fennica	0.88	0.04	0.87	0.47	0.218
Bombus vestalis	0.88	0.20	0.31	0.39	0.069
Andrena flavipes	0.88	0.08	0.39	0.04	0.139
Lasioglossum morio	0.87	0.10	0.44	0.14	0.079
Nomada fulvicornis	0.87	0.02	0.91	0.20	0.218
Andrena ovatula	0.87	0.05	0.32	0.14	0.198
Ancistrocerus nigricornis	0.86	0.06	0.80	0.42	0.208
Blennocampa phyllocolpa	0.86	0.08	0.95	0.30	0.198
Ectemnius cephalotes	0.85	0.11	0.16	0.22	0.119
Chelostoma campanularum	0.84	0.03	0.87	0.06	0.139
Cerceris ruficornis	0.83	0.08	0.46	0.46	0.089
Chrysis illigeri	0.83	0.00	0.97	0.45	0.218
Megachile versicolor	0.82	0.07	0.58	0.30	0.208
Elampus panzeri	0.81	0.01	0.97	0.69	0.099
Eriocampa ovata	0.79	0.13	0.85	0.27	0.149
Nomada sheppardana	0.79	0.07	0.74	0.08	0.069
Formica fusca	0.79	0.12	0.11	0.13	0.050
Neuroterus albipes	0.77	0.04	0.80	0.42	0.386
Halictus tumulorum	0.77	0.08	0.44	0.16	0.069
Nomada flavoguttata	0.75	0.08	0.64	0.20	0.119
Leptothorax nylanderi	0.75	0.01	0.96	0.88	0.248
Lasioglossum laevigatum	0.75	0.10	0.74	0.46	0.149
Melitta tricincta	0.74	0.02	0.92	0.43	0.089
Osmia caerulescens	0.73	0.09	0.53	0.23	0.079
Andrena clarkella	0.72	0.11	0.66	0.12	0.119
Abia nitens	0.71	0.10	0.92	0.00	0.059
Vespa crabro	0.71	0.14	0.66	0.35	0.139
Anthophora quadrimaculata	0.71	0.03	0.13	0.07	0.208
Panurgus calcaratus	0.71	0.01	0.90	0.28	0.267
Nomada ruficornis	0.70	0.11	0.73	0.16	0.178

Andrena semilaevis	0.69	0.07	0.16	0.17	0.198
Lasius fuliginosus	0.68	0.10	0.60	0.33	0.119
Abia fasciata	0.67	0.13	0.48	0.07	0.030
Tenthredo arcuata	0.67	0.19	0.39	0.05	0.010
Megachile centuncularis	0.67	0.08	0.25	0.05	0.158
Andrena thoracica	0.67	0.01	0.71	0.05	0.406
Colletes daviesanus	0.66	0.14	0.44	0.07	0.099
Crabro peltarius	0.65	0.04	0.81	0.22	0.139
Pimpla rufipes	0.65	0.10	0.88	0.27	0.099
Rhogogaster scalaris	0.63	0.10	0.70	0.26	0.079
Lindenius albilabris	0.62	0.14	0.44	0.25	0.139
Andrena marginata	0.62	0.00	0.96	0.00	0.337
Sphecodes gibbus	0.61	0.06	0.41	0.27	0.208
Hoplitis claviventris	0.61	0.11	0.46	0.28	0.089
Vespula germanica	0.61	0.15	0.19	0.19	0.099
Nomada flavopicta	0.60	0.07	0.47	0.29	0.099
Lasioglossum minutissimum	0.60	0.05	0.79	0.17	0.119
Lasioglossum puncticolle	0.60	0.02	0.84	0.34	0.238
Trypoxylon medium	0.60	0.05	0.71	0.24	0.218
Lasioglossum zonulum	0.58	0.01	0.88	0.47	0.149
Caliadurgus fasciatellus	0.58	0.05	0.70	0.41	0.149
Andrena humilis	0.58	0.09	0.29	0.14	0.168
Andrena alfkenella	0.57	0.11	0.39	0.48	0.119
Arachnospila minutula	0.57	0.04	0.48	0.31	0.257
Andrena carantonica	0.55	0.19	0.36	0.14	0.069
Nysson trimaculatus	0.55	0.11	0.67	0.39	0.109
Anthophora furcata	0.55	0.05	0.64	0.21	0.168
Lasioglossum leucopus	0.54	0.09	0.44	0.08	0.059
Macrophya annulata	0.54	0.07	0.94	0.75	0.139
Andrena denticulata	0.54	0.10	0.36	0.06	0.079
Colletes cunicularius	0.53	0.05	0.90	0.09	0.059
Neuroterus quercusbaccarum	0.53	0.22	0.52	0.00	0.050
Neuroterus numismalis	0.52	0.28	0.62	0.03	0.020
Mimesa equestris	0.52	0.16	0.53	0.35	0.059
Leptothorax acervorum	0.52	0.11	0.09	0.00	0.158
Crossocerus cetratus	0.52	0.22	0.40	0.23	0.079
Megachile leachella	0.52	0.04	0.50	0.11	0.158
Andrena helvola	0.50	0.07	0.78	0.15	0.218
Lasioglossum cupromicans	0.50	0.09	0.54	0.33	0.208
Hylaeus hyalinatus	0.49	0.07	0.47	0.23	0.139
Andrena fulva	0.49	0.27	0.26	0.21	0.030
Nomada flava	0.48	0.18	0.27	0.15	0.079
Hylaeus pictipes	0.47	0.01	0.94	0.58	0.267
Diodontus insidiosus	0.47	0.03	0.83	0.50	0.198
Athalia circularis	0.47	0.11	0.76	0.17	0.119
Sphecodes monilicornis	0.47	0.08	0.33	0.12	0.059

Cimbex femoratus	0.46	0.13	0.45	0.05	0.218
Allantus cinctus	0.46	0.00	0.96	0.89	0.347
Deuteragenia variegata	0.45	0.09	0.32	0.22	0.119
Oxybelus argentatus	0.45	0.03	0.71	0.44	0.139
Sphecodes puncticeps	0.45	0.19	0.15	0.22	0.089
Psenulus pallipes agg	0.45	0.09	0.56	0.03	0.198
Andrena bicolor	0.44	0.11	0.47	0.11	0.119
Priocnemis parvula	0.44	0.11	0.29	0.14	0.109
Tenthredo scrophulariae	0.43	0.04	0.97	0.61	0.168
Andricus lignicolus	0.43	0.16	0.54	0.26	0.129
Lasioglossum rufitarse	0.43	0.25	0.82	0.12	0.020
Bombus sylvestris	0.43	0.22	0.19	0.08	0.149
Andrena synadelpha	0.42	0.13	0.59	0.41	0.129
Andrena fuscipes	0.41	0.05	0.83	0.18	0.079
Myrmica schencki	0.41	0.07	0.58	0.52	0.089
Sphecodes crassus	0.41	0.17	0.26	0.38	0.069
Evagetes crassicornis	0.40	0.10	0.47	0.18	0.099
Bombus lapidarius	0.40	0.25	0.10	0.17	0.079
Andrena fulvago	0.40	0.02	0.84	0.42	0.218
Crossocerus nigritus	0.40	0.03	0.93	0.41	0.059
Anoplius viaticus	0.39	0.06	0.62	0.61	0.208
Epeolus cruciger	0.39	0.11	0.56	0.17	0.149
Ancistrocerus scoticus	0.38	0.02	0.48	0.17	0.436
Sapyga quinquepunctata	0.38	0.09	0.13	0.53	0.218
Eutomostethus ephippium	0.37	0.01	0.99	0.75	0.168
Crossocerus annulipes	0.37	0.05	0.84	0.32	0.178
Ceratina cyanea	0.37	0.00	0.96	0.59	0.178
Aporus unicolor	0.37	0.00	0.98	0.83	0.238
Athalia lugens	0.37	0.06	0.89	0.50	0.099
Megachile circumcincta	0.37	0.06	0.80	0.36	0.208
Cryptocheilus variipennis	0.35	0.02	0.31	0.04	0.317
Podalonia affinis	0.32	0.09	0.84	0.50	0.168
Macropis europaea	0.32	0.00	1.00	1.00	0.248
Vespula vulgaris	0.32	0.20	0.20	0.06	0.050
Dasypoda hirtipes	0.32	0.03	0.50	0.29	0.178
Coelioxys elongatus	0.32	0.06	0.64	0.02	0.188
Bombus pratorum	0.30	0.22	0.18	0.09	0.089
Entomognathus brevis	0.30	0.09	0.69	0.50	0.158
Argogorytes mystaceus	0.29	0.17	0.67	0.09	0.089
Microdynerus exilis	0.29	0.01	0.96	0.80	0.238
Gymnomerus laevipes	0.28	0.13	0.31	0.27	0.119
Bombus bohemicus	0.28	0.09	0.33	0.01	0.168
Ceropales maculata	0.28	0.01	0.79	0.00	0.406
Colletes fodiens	0.28	0.03	0.70	0.37	0.228
Oxybelus uniglumis	0.28	0.12	0.42	0.20	0.089
Methocha articulata	0.27	0.03	0.83	0.22	0.218

Pemphredon lugubris	0.27	0.08	0.73	0.21	0.168
Crossocerus quadrimaculatus	0.26	0.13	0.58	0.17	0.069
Hemichroa australis	0.25	0.01	1.00	0.76	0.257
Nesoselandria morio	0.25	0.03	0.59	0.28	0.396
Rhogogaster chlorosoma	0.25	0.02	0.99	0.00	0.089
Colletes succinctus	0.24	0.06	0.34	0.03	0.307
Crossocerus podagricus	0.24	0.17	0.37	0.28	0.149
Crabro scutellatus	0.24	0.02	0.88	0.80	0.208
Andrena angustior	0.23	0.02	0.61	0.02	0.356
Ectemnius sexcinctus	0.23	0.07	0.82	0.41	0.168
Dolerus eversmanni	0.22	0.06	0.85	0.79	0.228
Tenthredopsis coquebertii	0.22	0.05	0.71	0.09	0.238
Priocnemis hyalinata	0.22	0.03	0.91	0.56	0.119
Rhyssa persuasoria	0.22	0.07	0.29	0.11	0.069
Epeolus variegatus	0.20	0.05	0.46	0.15	0.198
Lasius umbratus	0.20	0.05	0.56	0.39	0.129
Andrena bucephala	0.20	0.20	0.38	0.33	0.040
Andrena tibialis	0.19	0.07	0.59	0.45	0.168
Smicromyrme rufipes	0.19	0.01	0.96	0.84	0.168
Hylaeus cornutus	0.18	0.02	0.81	0.14	0.089
Bombus humilis	0.17	0.04	0.52	0.32	0.396
Ammophila pubescens	0.17	0.02	0.93	0.76	0.188
Cladius brullei	0.17	0.04	0.89	0.52	0.168
Diplazon pectoratorius	0.17	0.00	1.00	0.92	0.248
Ichneumon suspiciosus	0.17	0.03	0.97	0.65	0.119
Chrysis ignita	0.17	0.25	0.41	0.28	0.059
Andrena minutula	0.17	0.19	0.33	0.08	0.119
Nomada baccata	0.16	0.01	0.98	0.96	0.149
Harpactus tumidus	0.15	0.10	0.69	0.47	0.109
Arachnospila wesmaeli	0.15	0.07	0.74	0.71	0.208
Athalia liberta	0.15	0.10	0.57	0.27	0.139
Lasioglossum pauperatum	0.15	0.00	1.00	1.00	1.000
Tachysphex pompiliformis	0.14	0.11	0.29	0.21	0.139
Dolichovespula sylvestris	0.14	0.10	0.26	0.02	0.317
Arachnospila anceps	0.13	0.04	0.51	0.13	0.079
Sphecodes hyalinatus	0.12	0.11	0.91	0.20	0.089
Pseudomalus violaceus	0.12	0.01	0.90	0.86	0.366
Passaloecus gracilis	0.12	0.04	0.73	0.36	0.129
Lasius niger	0.10	0.14	0.25	0.08	0.129
Andrena chrysosceles	0.10	0.21	0.44	0.29	0.079
Priocnemis schioedtei	0.09	0.04	0.40	0.26	0.267
Lasioglossum smeathmanellum	0.09	0.04	0.34	0.18	0.188
Mellinus arvensis	0.09	0.15	0.35	0.09	0.079
Formica sanguinea	0.09	0.05	0.92	0.03	0.158
Pontania proxima	0.08	0.09	0.80	0.33	0.149
Tenthredo mesomela	0.08	0.19	0.56	0.24	0.168

Lasius flavus	0.08	0.12	0.21	0.19	0.168
Arachnospila spissa	0.06	0.13	0.54	0.28	0.069
Crossocerus dimidiatus	0.05	0.11	0.72	0.03	0.050
Xiphydria camelus	0.05	0.09	0.85	0.79	0.139
Ammophila sabulosa	0.05	0.09	0.35	0.15	0.158
Coelioxys rufescens	0.04	0.03	0.75	0.12	0.188
Andrena tarsata	0.03	0.04	0.60	0.07	0.079
Ectemnius dives	0.03	0.01	0.83	0.41	0.218
Oxybelus mandibularis	0.03	0.01	0.82	0.56	0.297
Andrena pilipes	0.02	0.02	0.52	0.78	0.119
Mimumesa dahlbomi	0.01	0.17	0.19	0.23	0.149
Crossocerus megacephalus	0.00	0.04	0.69	0.13	0.248
Crossocerus elongatulus	0.00	0.04	0.26	0.12	0.287
Crossocerus wesmaeli	-0.01	0.03	0.80	0.40	0.178
Andrena fucata	-0.02	0.10	0.42	0.07	0.188
Dolerus aericeps	-0.02	0.08	0.71	0.32	0.208
Sphecodes longulus	-0.02	0.08	0.56	0.48	0.129
Crossocerus capitosus	-0.02	0.12	0.39	0.38	0.168
Ancistrocerus trifasciatus	-0.02	0.17	0.57	0.19	0.099
Cladius compressicornis	-0.03	0.05	0.92	0.72	0.248
Nematus ribesii	-0.03	0.10	0.23	0.22	0.119
Monophadnus pallescens	-0.03	0.07	0.78	0.41	0.257
Omalus aeneus	-0.03	0.00	1.00	1.00	0.436
Bombus barbutellus	-0.03	0.09	0.47	0.16	0.198
Arge gracilicornis	-0.03	0.02	0.64	0.44	0.485
Chrysis ruddii	-0.03	0.02	0.63	0.00	0.347
Osmia aurulenta	-0.04	0.03	0.63	0.21	0.089
Gorytes quadrifasciatus	-0.04	0.09	0.70	0.38	0.178
Hylaeus brevicornis	-0.05	0.08	0.57	0.16	0.158
Eutomostethus luteiventris	-0.05	0.09	0.85	0.25	0.257
Biorhiza pallida	-0.05	0.15	0.24	0.19	0.218
Eucera longicornis	-0.05	0.05	0.49	0.48	0.238
Hylaeus communis	-0.06	0.15	0.50	0.19	0.119
Ectemnius cavifrons	-0.07	0.11	0.65	0.30	0.149
Bombus pascuorum	-0.08	0.25	0.05	0.06	0.089
Sphecodes miniatus	-0.08	0.02	0.50	0.41	0.248
Lasioglossum fulvicorne	-0.11	0.19	0.48	0.19	0.139
Pompilus cinereus	-0.12	0.10	0.59	0.30	0.119
Andrena congruens	-0.12	0.02	0.96	0.25	0.119
Bethylus cephalotes	-0.12	0.06	0.82	0.37	0.139
Tenthredo atra	-0.12	0.07	0.58	0.36	0.158
Arge ustulata	-0.12	0.08	0.41	0.24	0.228
Lasius mixtus	-0.12	0.02	0.74	0.52	0.099
Dolerus cothurnatus	-0.13	0.03	0.92	0.67	0.287
Nomada striata	-0.13	0.14	0.42	0.24	0.109
Andrena lapponica	-0.14	0.18	0.76	0.02	0.030

Andricus solitarius	-0.15	0.00	1.00	1.00	1.000
Lissonota coracina	-0.15	0.03	0.99	0.06	0.040
Lonchodryinus ruficornis	-0.15	0.10	0.82	0.25	0.069
Tenthredo temula	-0.15	0.14	0.33	0.44	0.168
Diodontus tristis	-0.16	0.10	0.62	0.31	0.109
Coelioxys inermis	-0.17	0.07	0.62	0.28	0.168
Tetramorium caespitum	-0.17	0.04	0.36	0.46	0.099
Andrena barbilabris	-0.18	0.09	0.60	0.05	0.168
Pemphredon inornata	-0.20	0.06	0.73	0.25	0.198
Stelis ornatula	-0.21	0.12	0.57	0.47	0.198
Dolerus madidus	-0.21	0.08	0.68	0.37	0.109
Nematus lucidus	-0.21	0.06	0.68	0.67	0.139
Pemphredon morio	-0.21	0.07	0.78	0.41	0.158
Phyllocolpa leucosticta	-0.21	0.15	0.90	0.15	0.010
Podalonia hirsuta	-0.22	0.07	0.76	0.61	0.158
Aglaostigma aucupariae	-0.23	0.13	0.70	0.66	0.139
Hylaeus pectoralis	-0.23	0.07	0.71	0.79	0.158
Sphecodes pellucidus	-0.25	0.04	0.72	0.33	0.158
Nomada marshamella	-0.26	0.17	0.44	0.11	0.089
Lasioglossum fratellum	-0.26	0.25	0.47	0.02	0.059
Megachile maritima	-0.28	0.03	0.74	0.49	0.347
Crossocerus binotatus	-0.28	0.03	0.95	0.00	0.129
Coelioxys mandibularis	-0.29	0.05	0.84	0.57	0.030
Lasioglossum xanthopus	-0.29	0.01	0.98	0.36	0.139
Lasioglossum parvulum	-0.29	0.09	0.69	0.31	0.129
Abia lonicerae	-0.29	0.02	0.95	0.91	0.317
Hoplocampa crataegi	-0.29	0.04	0.91	0.71	0.238
Itoplectis alternans	-0.29	0.03	0.41	0.87	0.267
Ametastegia equiseti	-0.32	0.04	0.66	0.43	0.297
Athalia bicolor	-0.32	0.12	0.84	0.55	0.109
Bombus monticola	-0.34	0.20	0.13	0.00	0.010
Stigmus solskyi	-0.34	0.08	0.64	0.49	0.168
Lasioglossum punctatissimum	-0.34	0.04	0.54	0.23	0.188
Trypoxylon attenuatum	-0.34	0.16	0.46	0.26	0.129
Myrmica lobicornis	-0.36	0.21	0.74	0.00	0.040
Andricus curvator	-0.36	0.17	0.51	0.25	0.069
Lestiphorus bicinctus	-0.36	0.01	0.88	0.00	0.198
Ametastegia carpini	-0.37	0.00	1.00	1.00	0.356
Didineis lunicornis	-0.37	0.05	0.90	0.77	0.079
Andrena labialis	-0.37	0.14	0.26	0.16	0.149
Lasioglossum villosulum	-0.39	0.05	0.42	0.23	0.149
Andrena argentata	-0.39	0.01	0.97	0.94	0.218
Andrena varians	-0.40	0.03	0.84	0.29	0.168
Dolerus ferrugatus	-0.40	0.12	0.57	0.24	0.089
Chrysis viridula	-0.40	0.16	0.51	0.04	0.040
Colletes marginatus	-0.40	0.01	0.78	0.59	0.198

Passaloecus corniger	-0.41	0.12	0.62	0.50	0.139
Hylaeus confusus	-0.43	0.10	0.58	0.14	0.069
Myrmica ruginodis	-0.44	0.12	0.27	0.00	0.119
Euura viminalis	-0.44	0.08	0.28	0.07	0.218
Andricus inflator	-0.45	0.02	0.96	0.67	0.248
Priocnemis pusilla	-0.48	0.12	0.25	0.42	0.109
Evagetes dubius	-0.48	0.00	0.93	0.68	0.297
Dolerus vestigialis	-0.49	0.13	0.74	0.40	0.158
Odynerus melanocephalus	-0.49	0.07	0.30	0.09	0.178
Andricus foecundatrix	-0.49	0.14	0.30	0.17	0.139
Tenthredo brevicornis	-0.49	0.07	0.77	0.19	0.158
Trypoxylon figulus agg	-0.49	0.17	0.71	0.10	0.099
Colletes similis	-0.50	0.01	0.40	0.05	0.356
Dolerus varispinus	-0.52	0.14	0.42	0.07	0.089
Bombus lucorum	-0.52	0.20	0.23	0.02	0.119
Strongylogaster multifasciata	-0.52	0.09	0.62	0.15	0.119
Tenthredo livida	-0.53	0.35	0.38	0.15	0.010
Cephus spinipes	-0.53	0.03	0.97	0.87	0.139
Diodontus luperus	-0.54	0.13	0.52	0.30	0.099
Bombus distinguendus	-0.55	0.04	0.25	0.47	0.218
Trypoxylon clavicerum	-0.56	0.12	0.44	0.32	0.099
Andrena simillima	-0.58	0.03	0.61	0.09	0.267
Itoplectis maculator	-0.58	0.07	0.17	0.56	0.248
Phaneroserphus calcar	-0.58	0.07	0.98	0.09	0.050
Rhopalum clavipes	-0.59	0.05	0.68	0.12	0.109
Priocnemis exaltatus	-0.59	0.11	0.12	0.38	0.139
Tachysphex unicolor	-0.59	0.00	0.98	0.92	0.485
Pachyprotasis rapae	-0.59	0.11	0.46	0.16	0.158
Urocerus gigas	-0.62	0.12	0.23	0.03	0.287
Dolerus picipes	-0.63	0.12	0.38	0.34	0.079
Macrophya duodecimpunctata	-0.63	0.13	0.68	0.42	0.079
Alloxysta victrix	-0.63	0.08	0.41	0.78	0.119
Rhopalum coarctatum	-0.63	0.07	0.51	0.43	0.158
Mutilla europaea	-0.64	0.03	0.53	0.27	0.307
Crossocerus exiguus	-0.66	0.00	1.00	1.00	1.000
Monsoma pulveratum	-0.66	0.01	0.99	0.00	0.040
Nematus papillosus	-0.66	0.15	0.98	0.52	0.050
Dolerus nigratus	-0.66	0.06	0.54	0.56	0.238
Andricus kollari	-0.68	0.20	0.18	0.25	0.059
Dolerus germanicus	-0.69	0.13	0.85	0.24	0.119
Halictus confusus	-0.69	0.00	0.93	0.68	0.208
Formica cunicularia	-0.69	0.06	0.68	0.68	0.198
Dryudella pinguis	-0.69	0.11	0.51	0.36	0.099
Birka cinereipes	-0.69	0.06	0.96	0.73	0.119
Cladius pilicornis	-0.69	0.05	0.77	0.78	0.238
Euura clitellata	-0.69	0.18	0.94	0.18	0.050

Passaloecus clypealis	-0.69	0.00	1.00	0.97	0.099
Priocnemis confusor	-0.69	0.03	0.76	0.15	0.178
Ectemnius lapidarius	-0.70	0.14	0.55	0.20	0.139
Crabro cribrarius	-0.71	0.13	0.31	0.07	0.139
Crossocerus ovalis	-0.73	0.14	0.58	0.33	0.069
Colletes halophilus	-0.73	0.02	0.84	0.42	0.248
Allantus calceatus	-0.73	0.04	0.74	0.68	0.218
Myrmica rubra	-0.74	0.20	0.23	0.30	0.119
Bombus hortorum	-0.74	0.20	0.07	0.03	0.079
Formica lemani	-0.75	0.22	0.23	0.00	0.020
Platycampus luridiventris	-0.78	0.18	0.61	0.00	0.059
Bombus sylvarum	-0.78	0.01	0.77	0.71	0.505
Passaloecus singularis	-0.78	0.13	0.61	0.26	0.109
Miscophus concolor	-0.79	0.00	1.00	1.00	1.000
Pachynematus kirbyi	-0.82	0.10	0.92	0.05	0.050
Myrmosa atra	-0.83	0.13	0.30	0.22	0.059
Athalia cordata	-0.85	0.10	0.46	0.25	0.158
Bombus ruderatus	-0.85	0.15	0.19	0.49	0.099
Halidamia affinis	-0.86	0.10	0.87	0.66	0.089
Mimumesa littoralis	-0.86	0.06	0.84	0.69	0.040
Panurgus banksianus	-0.88	0.03	0.62	0.49	0.267
Agrypon flaveolatum	-0.92	0.00	1.00	1.00	1.000
Lasioglossum semilucens	-0.92	0.04	0.90	0.79	0.099
Pristiphora pallidiventris	-0.92	0.11	0.99	0.00	0.040
Trichiosoma lucorum	-0.92	0.16	0.64	0.00	0.030
Omalus puncticollis	-0.92	0.03	0.50	0.25	0.267
Melitta leporina	-0.94	0.03	0.57	0.31	0.149
Alomya debellator	-0.94	0.05	0.95	0.29	0.109
Dolerus brevicornis	-0.94	0.01	0.99	0.54	0.198
Nomada obtusifrons	-0.94	0.10	0.69	0.13	0.069
Bethylus fuscicornis	-0.94	0.03	0.94	0.64	0.109
Osmia parietina	-0.94	0.21	0.21	0.60	0.020
Rhogogaster punctulata	-0.94	0.25	0.54	0.16	0.040
Pemphredon lethifer	-0.97	0.07	0.51	0.13	0.168
Formica rufa	-0.98	0.28	0.18	0.34	0.040
Tenthredo maculata	-0.99	0.20	0.65	0.29	0.069
Andrena coitana	-1.01	0.15	0.58	0.03	0.059
Myrmica sabuleti	-1.02	0.13	0.08	0.22	0.109
Ancistrocerus parietinus	-1.03	0.12	0.25	0.12	0.109
Psenulus concolor	-1.03	0.06	0.84	0.46	0.139
Aneugmenus padi	-1.03	0.12	0.16	0.28	0.149
Lasioglossum nitidiusculum	-1.03	0.15	0.20	0.19	0.109
Selandria serva	-1.06	0.18	0.41	0.32	0.109
Lasius alienus	-1.07	0.03	0.45	0.18	0.198
Formica lugubris	-1.13	0.15	0.06	0.03	0.030
Diodontus minutus	-1.14	0.16	0.46	0.57	0.059

Myrmica scabrinodis	-1.14	0.09	0.36	0.00	0.168
Cynips longiventris	-1.14	0.03	0.77	0.51	0.356
Hylaeus incongruus	-1.14	0.11	0.84	0.73	0.188
Pachynematus obductus	-1.14	0.04	0.97	0.73	0.158
Crossocerus styrius	-1.14	0.01	0.97	0.79	0.267
Dolerus haematodes	-1.14	0.06	0.82	0.59	0.149
Crossocerus tarsatus	-1.15	0.12	0.33	0.06	0.129
Empria tridens	-1.16	0.08	0.71	0.44	0.119
Mimesa bruxellensis	-1.16	0.02	0.76	0.59	0.198
Cladius pectinicornis	-1.16	0.06	0.95	0.36	0.149
Arge melanochra	-1.16	0.00	1.00	1.00	1.000
Spilomena enslini	-1.16	0.00	0.97	0.86	0.238
Arachnospila trivialis	-1.19	0.13	0.50	0.57	0.109
Andrena nigriceps	-1.19	0.02	0.63	0.48	0.158
Ectemnius ruficornis	-1.21	0.05	0.53	0.11	0.307
Euura obducta	-1.22	0.00	1.00	1.00	1.000
Vespula rufa	-1.23	0.11	0.24	0.01	0.198
Crossocerus varus	-1.26	0.09	0.45	0.04	0.208
Bombus campestris	-1.28	0.16	0.30	0.21	0.139
Calameuta pallipes	-1.30	0.15	0.84	0.47	0.099
Pristiphora armata	-1.32	0.02	0.94	0.90	0.297
Tenthredopsis litterata	-1.33	0.13	0.27	0.30	0.079
Diplazon tetragonus	-1.35	0.00	1.00	0.99	0.149
Cladius ulmi	-1.36	0.01	0.96	0.95	0.287
Nomada roberjeotiana	-1.36	0.02	0.39	0.37	0.168
Pamphilius hortorum	-1.36	0.02	0.96	0.92	0.396
Scambus nigricans	-1.36	0.13	0.61	0.10	0.109
Tenthredo olivacea	-1.36	0.09	0.85	0.02	0.129
Tomostethus nigritus	-1.36	0.02	0.97	0.95	0.178
Stromboceros delicatulus	-1.38	0.14	0.34	0.16	0.059
Passaloecus eremita	-1.40	0.01	0.99	0.82	0.149
Aglaostigma fulvipes	-1.43	0.17	0.71	0.30	0.188
Crossocerus walkeri	-1.48	0.07	0.73	0.25	0.158
Tenthredo notha	-1.50	0.15	0.29	0.17	0.119
Cynips divisa	-1.53	0.11	0.44	0.24	0.248
Euura oligospila	-1.54	0.02	0.95	0.79	0.267
Bombus muscorum	-1.58	0.08	0.21	0.02	0.099
Dolichovespula norwegica	-1.59	0.13	0.31	0.00	0.149
Tenthredo colon	-1.61	0.08	0.57	0.21	0.228
Euura myosotidis	-1.67	0.03	0.85	0.62	0.257
Allantus cingulatus	-1.68	0.00	1.00	1.00	1.000
Neodiprion sertifer	-1.68	0.10	0.44	0.08	0.208
Passaloecus insignis	-1.71	0.06	0.84	0.37	0.139
Euura viridis	-1.72	0.06	0.71	0.10	0.129
Pamphilius sylvaticus	-1.72	0.00	0.93	0.97	0.416
Pachynematus clitellatus	-1.73	0.09	0.44	0.13	0.168

Cratichneumon culex	-1.73	0.01	1.00	0.99	0.109
Hartigia xanthostoma	-1.73	0.03	1.00	0.94	0.208
Osmia pilicornis	-1.73	0.00	1.00	1.00	1.000
Tenthredo schaefferi	-1.73	0.00	0.99	0.99	0.426
Trigonaspis megaptera	-1.73	0.08	0.59	0.19	0.050
Tenthredopsis nassata	-1.79	0.16	0.41	0.08	0.079
Andrena falsifica	-1.81	0.08	0.51	0.60	0.149
Dyspetes arrogator	-1.81	0.00	1.00	1.00	1.000
Spilomena troglodytes	-1.83	0.11	0.79	0.60	0.099
Dolerus niger	-1.85	0.12	0.28	0.18	0.129
Dolerus gonager	-1.88	0.16	0.71	0.04	0.030
Ametastegia glabrata	-1.90	0.14	0.78	0.50	0.129
Dolerus puncticollis	-1.96	0.12	0.97	0.62	0.099
Euura lichtwardti	-1.96	0.07	0.82	0.65	0.129
Dolerus fumosus	-2.01	0.11	0.65	0.25	0.089
Dolerus aeneus	-2.21	0.15	0.22	0.00	0.089
Bombus ruderarius	-2.23	0.07	0.71	0.44	0.149
Cephus nigrinus	-2.24	0.10	0.95	0.47	0.089
Spilomena differens	-2.24	0.00	1.00	1.00	1.000
Vespula austriaca	-2.36	0.13	0.20	0.10	0.059
Anteon pubicorne	-2.49	0.27	0.55	0.35	0.040
Macrophya ribis	-2.50	0.02	0.99	0.94	0.228
Macrophya albicincta	-2.99	0.01	0.98	0.95	0.257
Formica aquilonia	-5.55	0.10	0.18	0.03	0.040

Appendix C: Projected distribution change

Projected distribution change of Odonatan and Hymenopteran study species by 2040 under an RCP8.5 climate change scenario according to ensemble species distribution models.

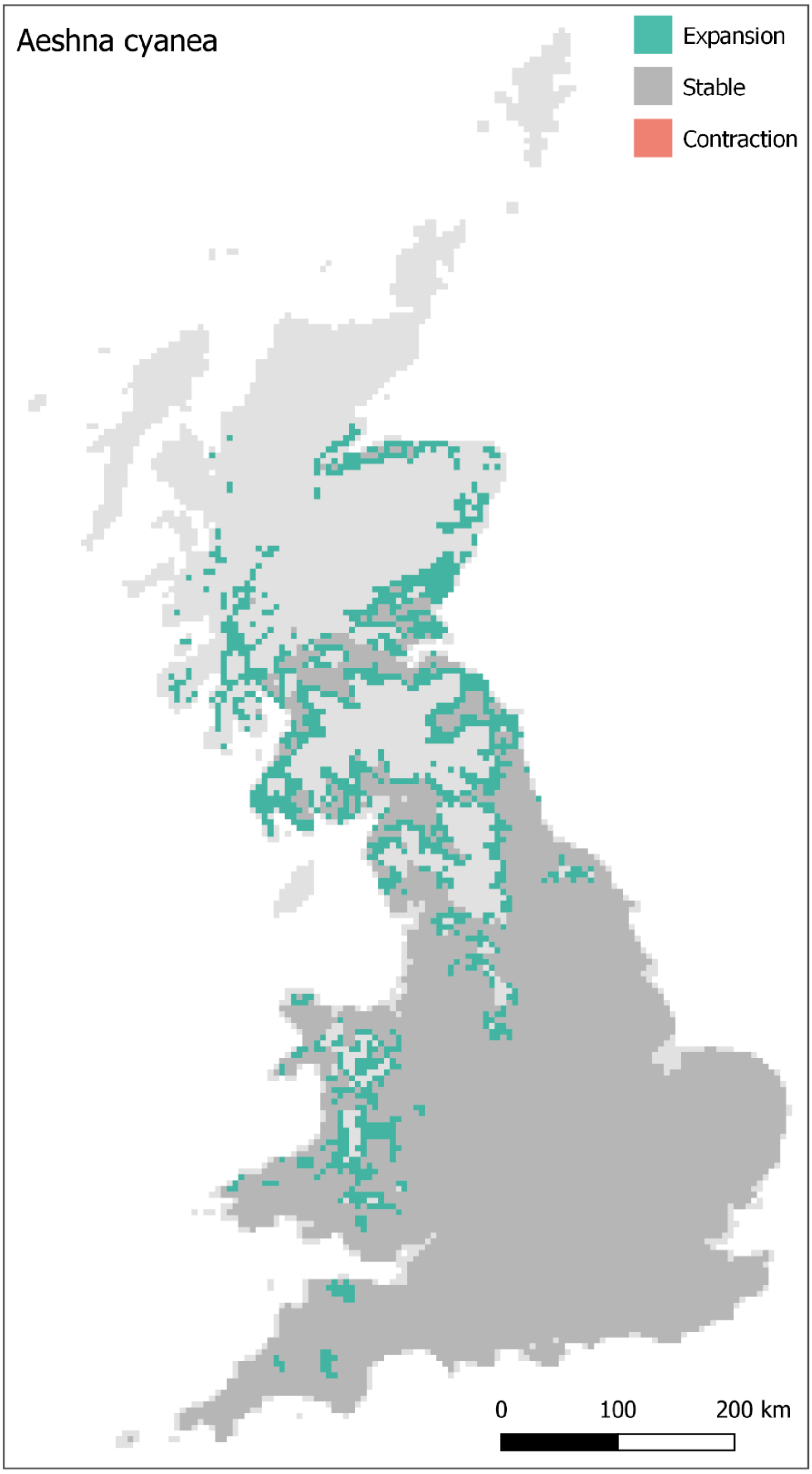

Figure C.1. Projected distribution change under climate change by 2040 of the Southern Hawker dragonfly (Aeshna cyanea).

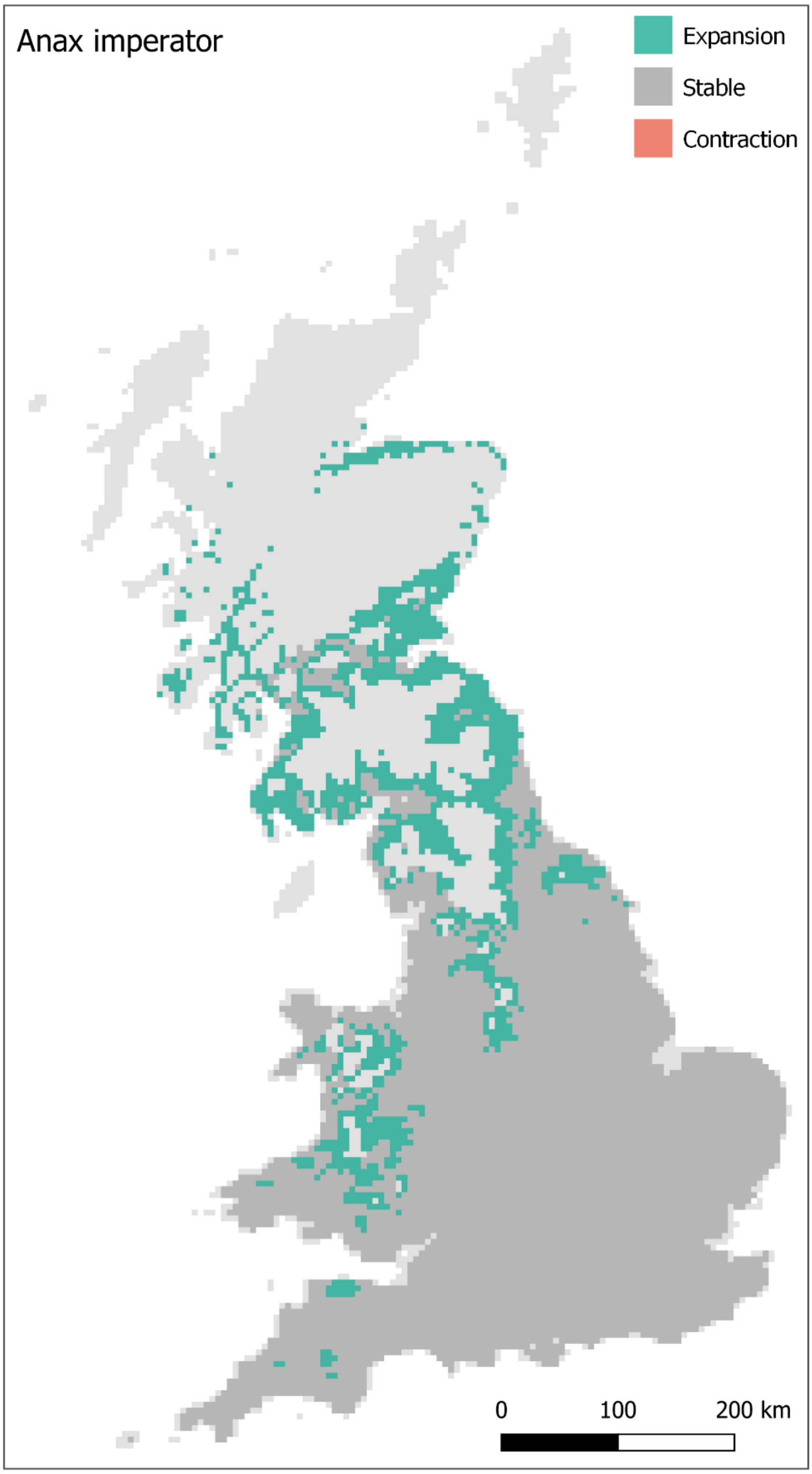

Figure C.2. Projected distribution change under climate change by 2040 of the Emperor Dragonfly (Anax imperator).

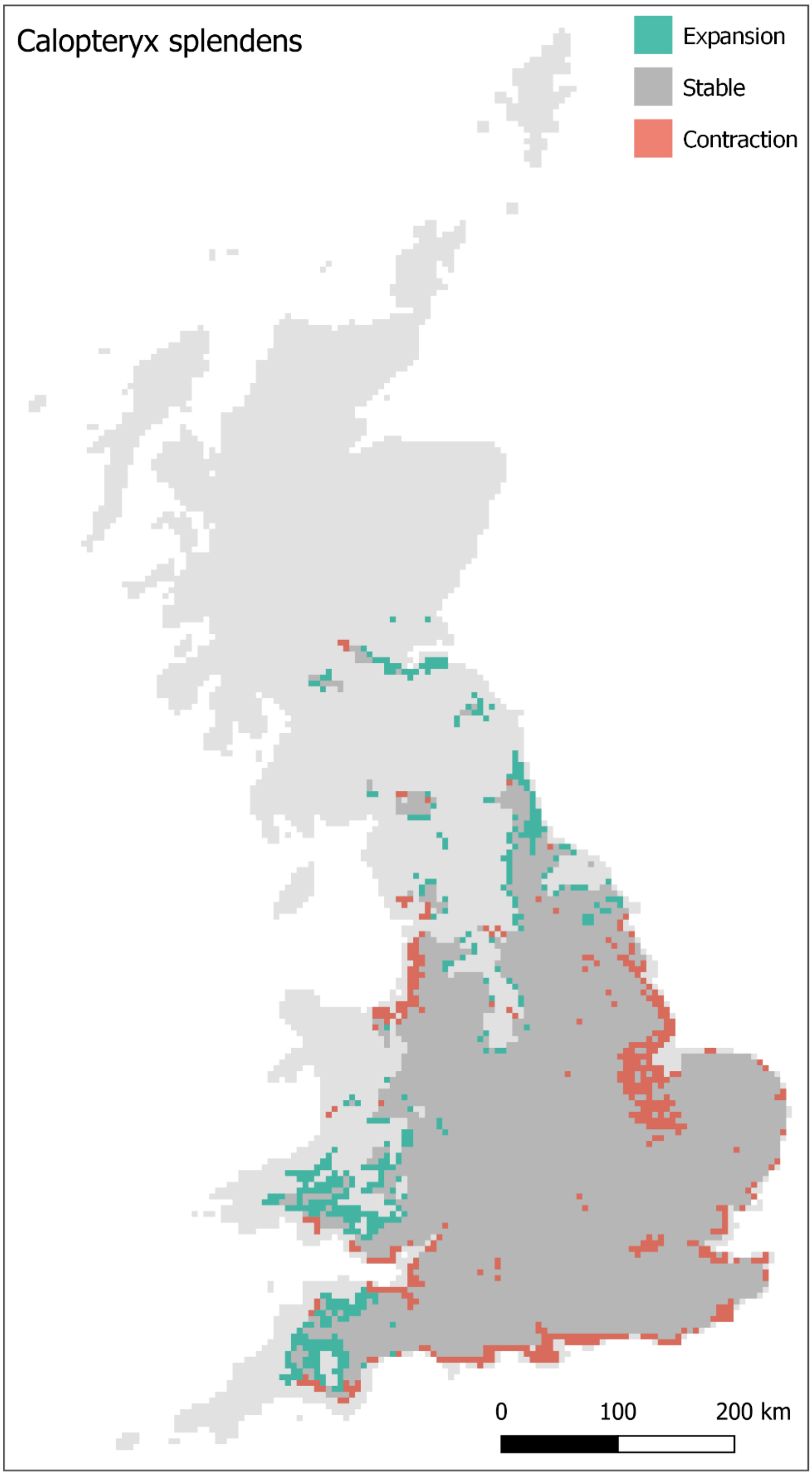

Figure C.3. Projected distribution change under climate change by 2040 of the Banded Demoiselle damselfly (Calopteryx splendens).

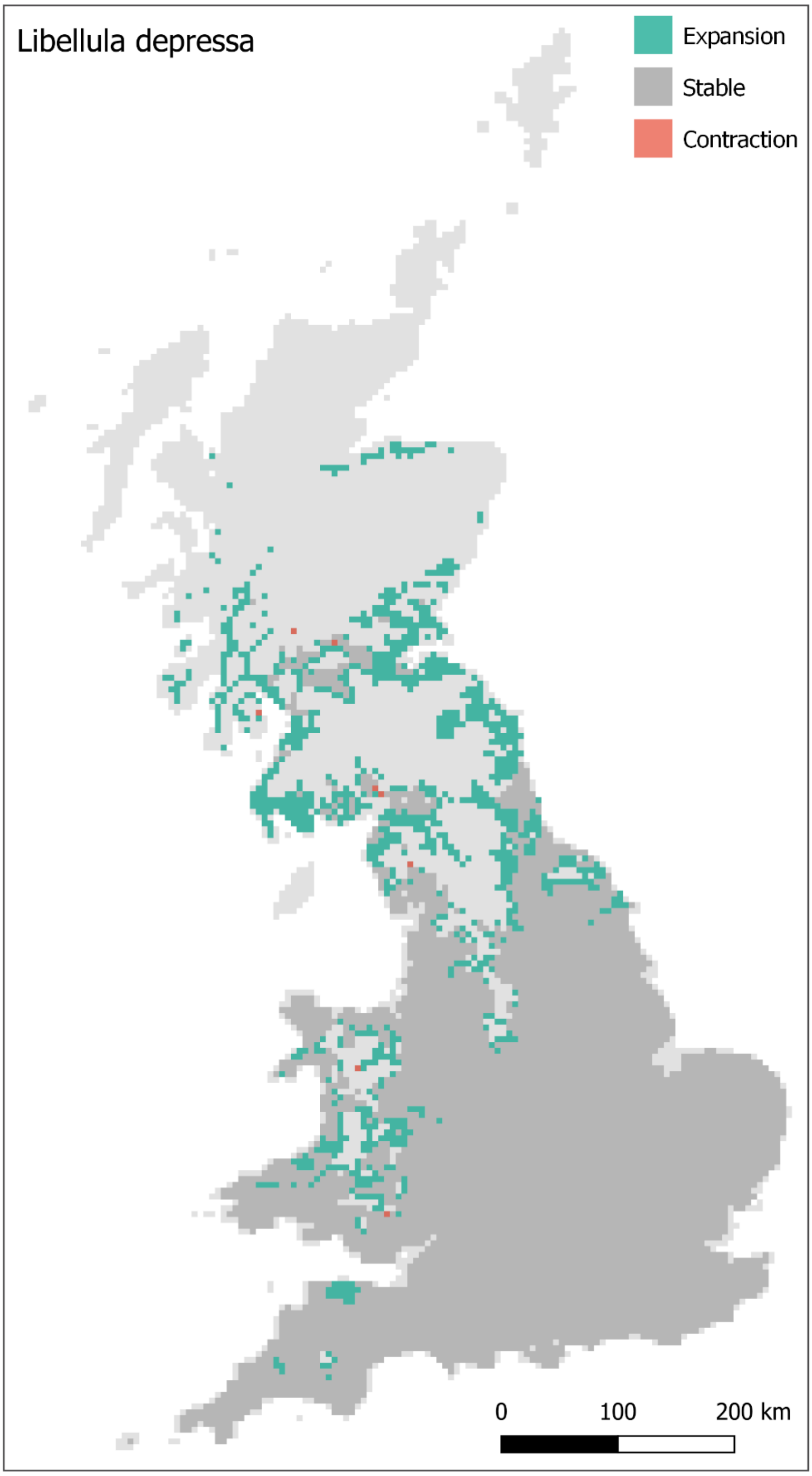

Figure C.4. Projected distribution change under climate change by 2040 of the Broad-bodied Chaser dragonfly (Libellula depressa).

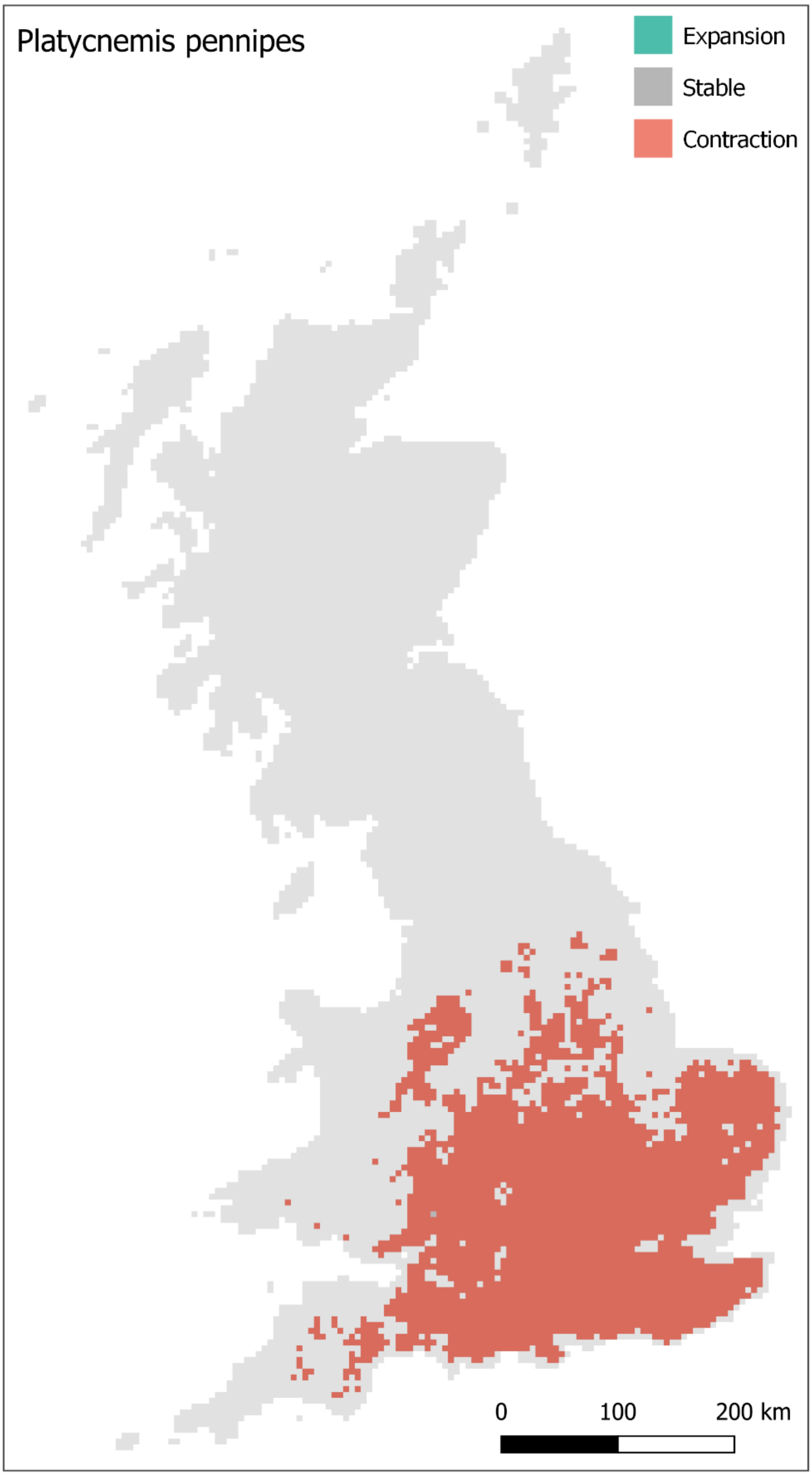

Figure C.5. Projected distribution change under climate change by 2040 of the White-legged Damselfly (Platycnemis pennipes).

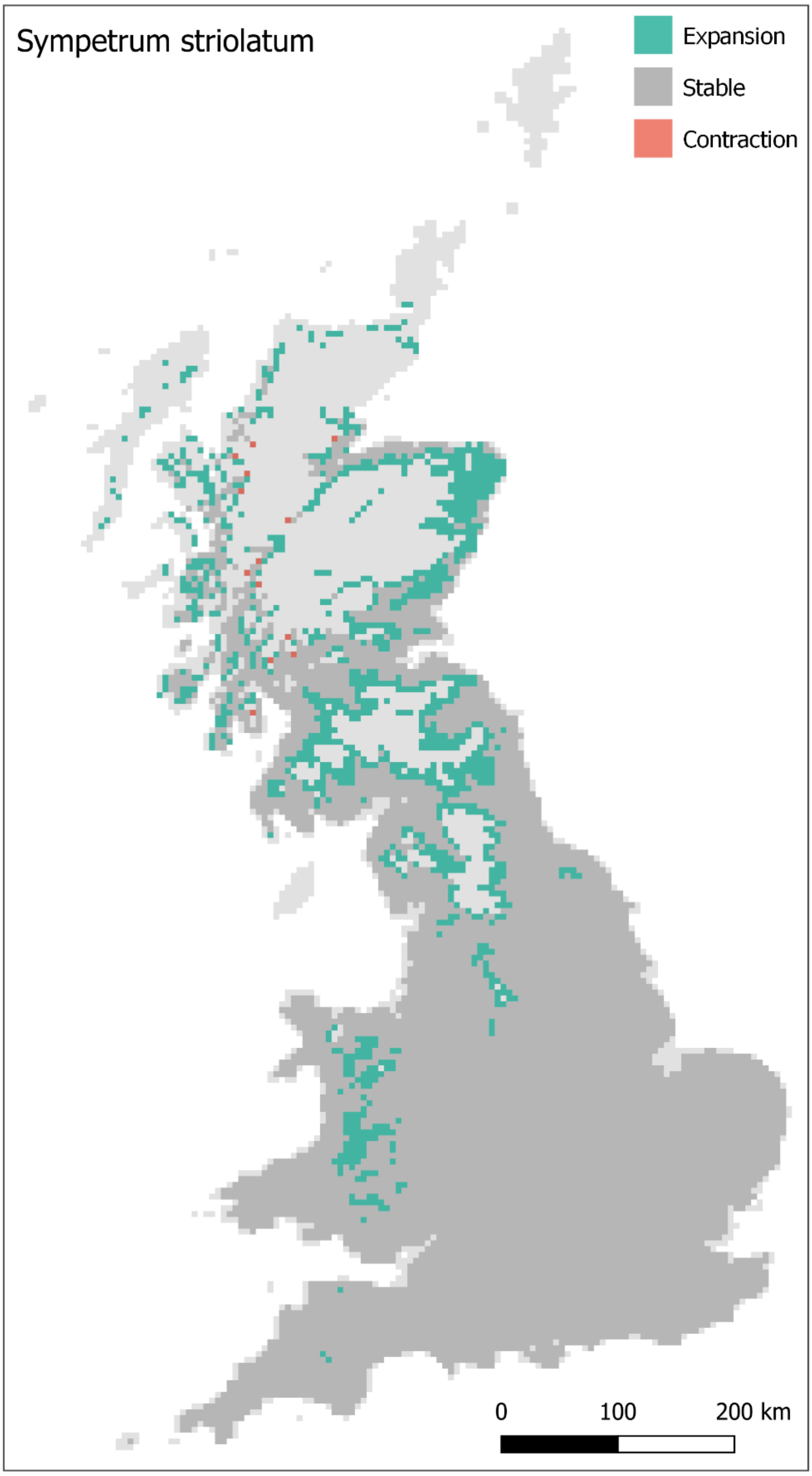

Figure C.6. Projected distribution change under climate change by 2040 of the Common Darter dragonfly (Sympetrum striolatum).

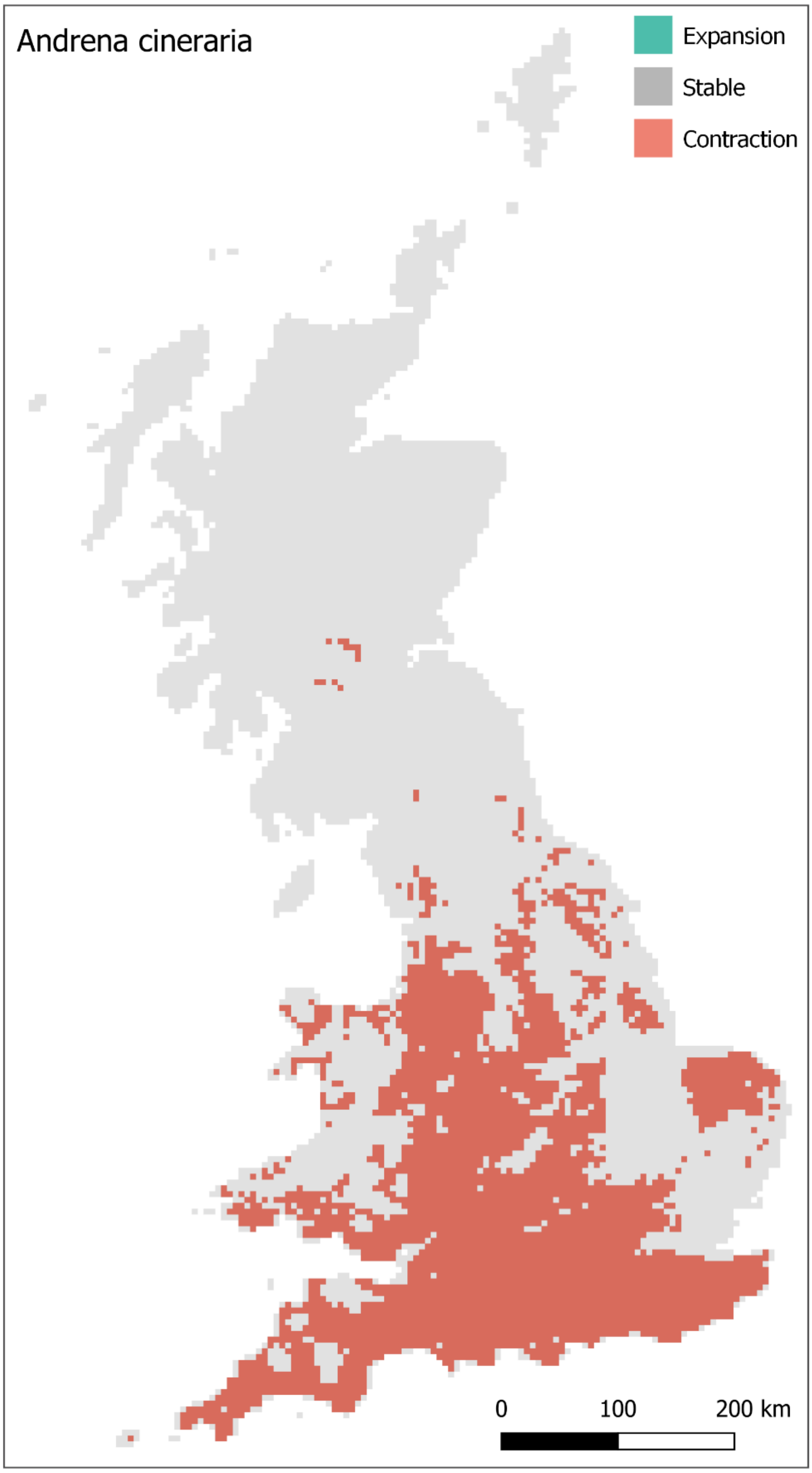

Figure C.7. Projected distribution change under climate change by 2040 of the Ashy Mining Bee (Andrena cineraria).

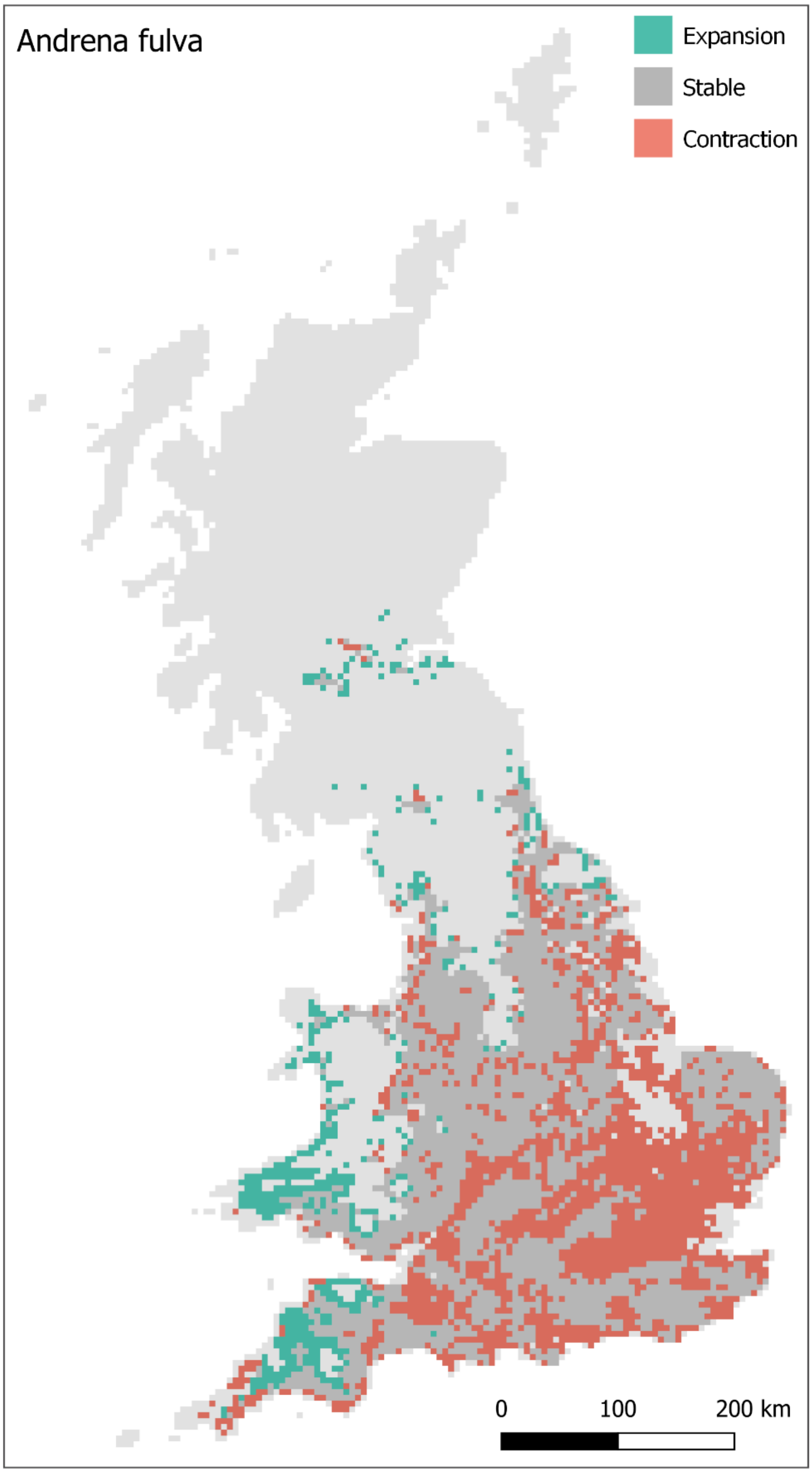

Figure C.8. Projected distribution change under climate change by 2040 of the Tawny Mining Bee (Andrena fulva).

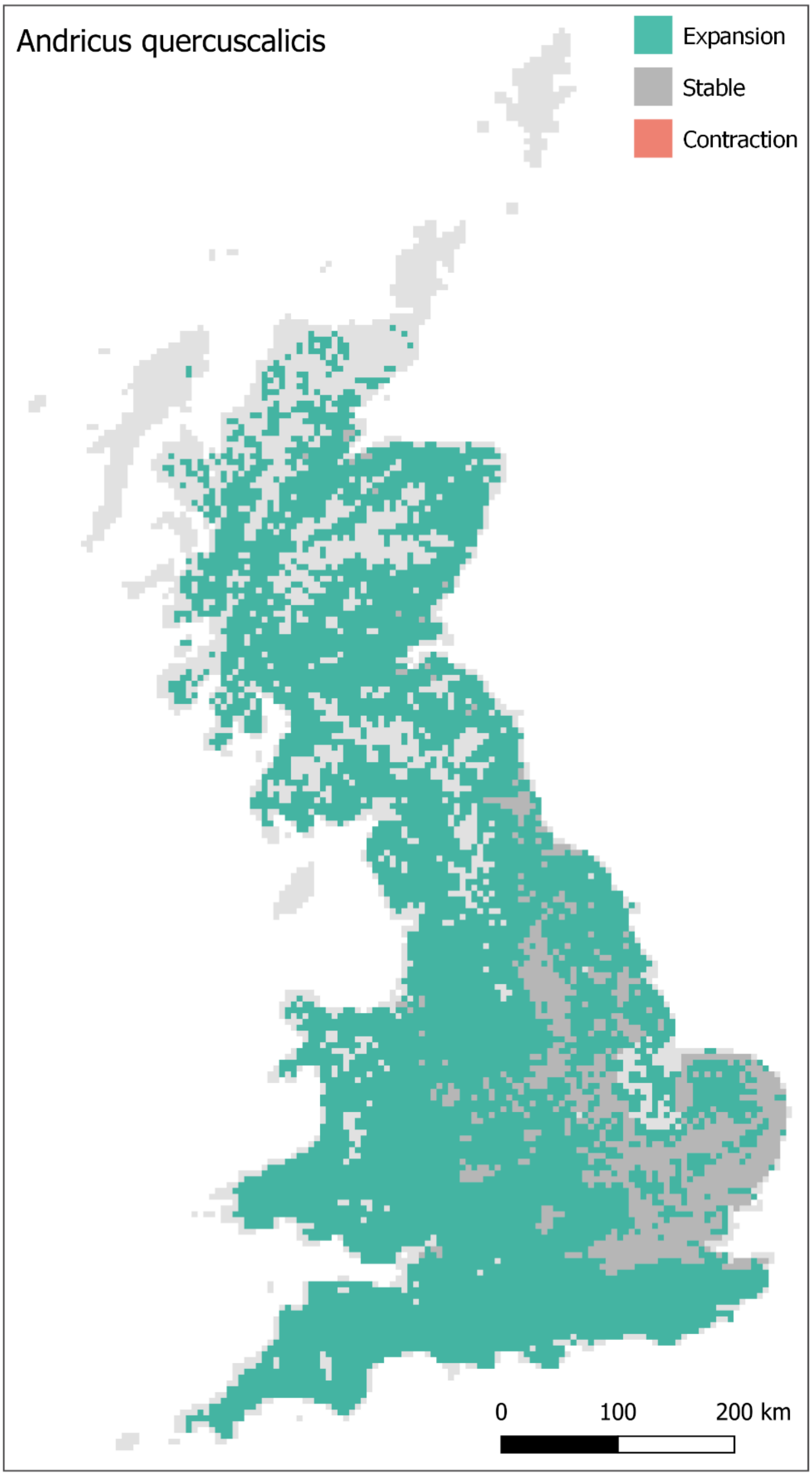

Figure C.9. Projected distribution change under climate change by 2040 of the Knopper Gall Wasp (Andricus quercuscalicis).

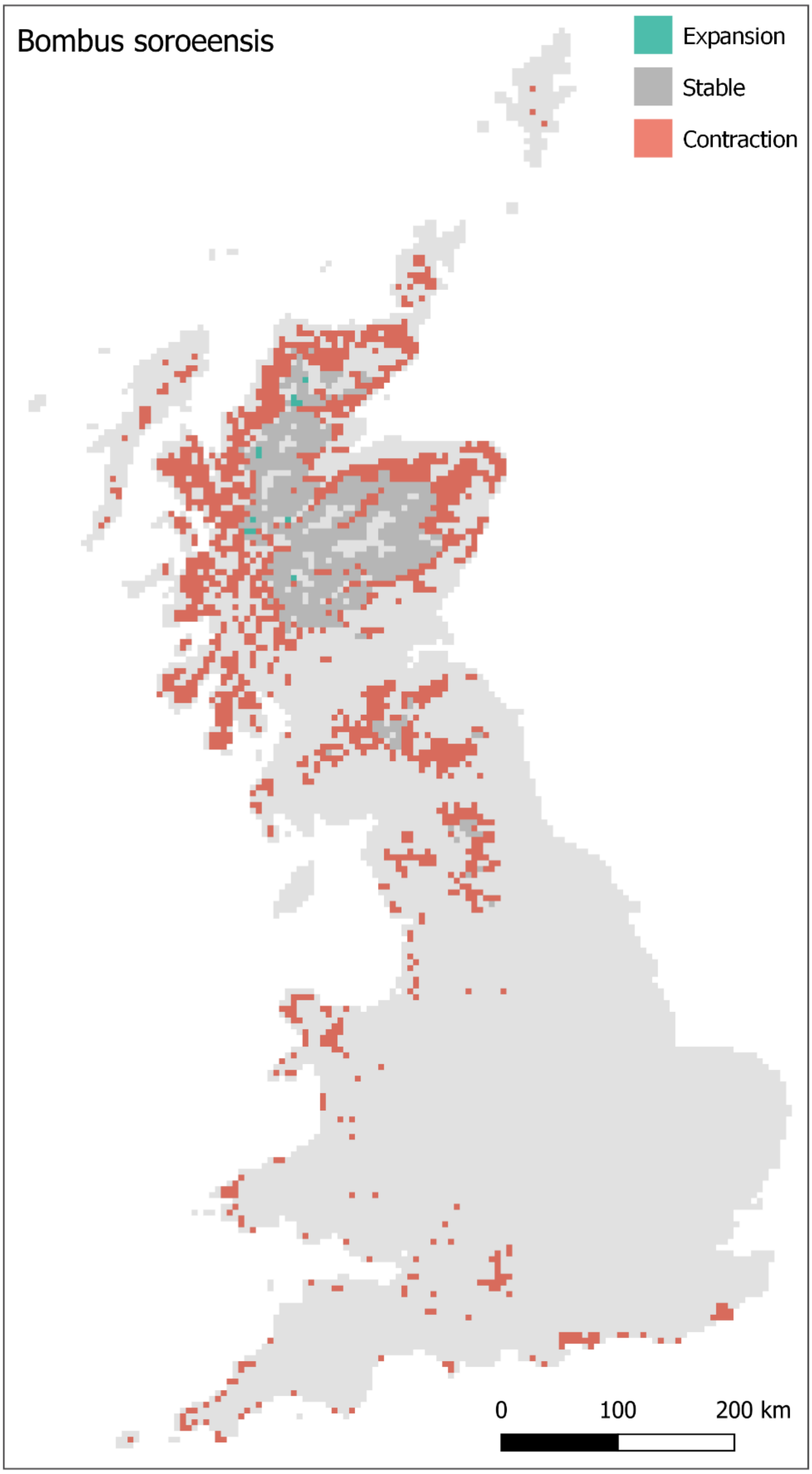

Figure C.10. Projected distribution change under climate change by 2040 of the Broken-belted Bumblebee (Bombus soroeensis).

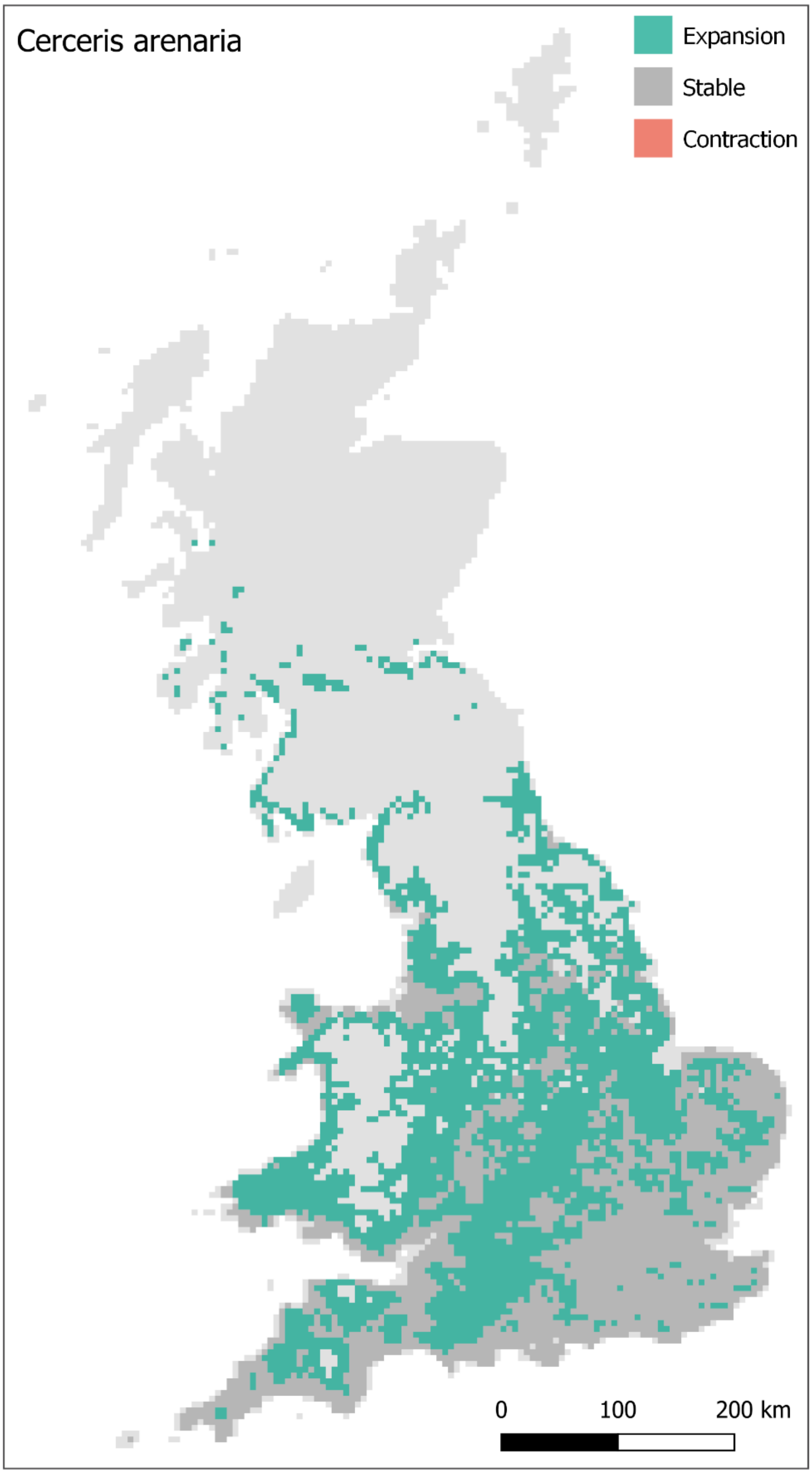

Figure C.11. Projected distribution change under climate change by 2040 of the Sand Tailed Digger Wasp (Cerceris arenaria).

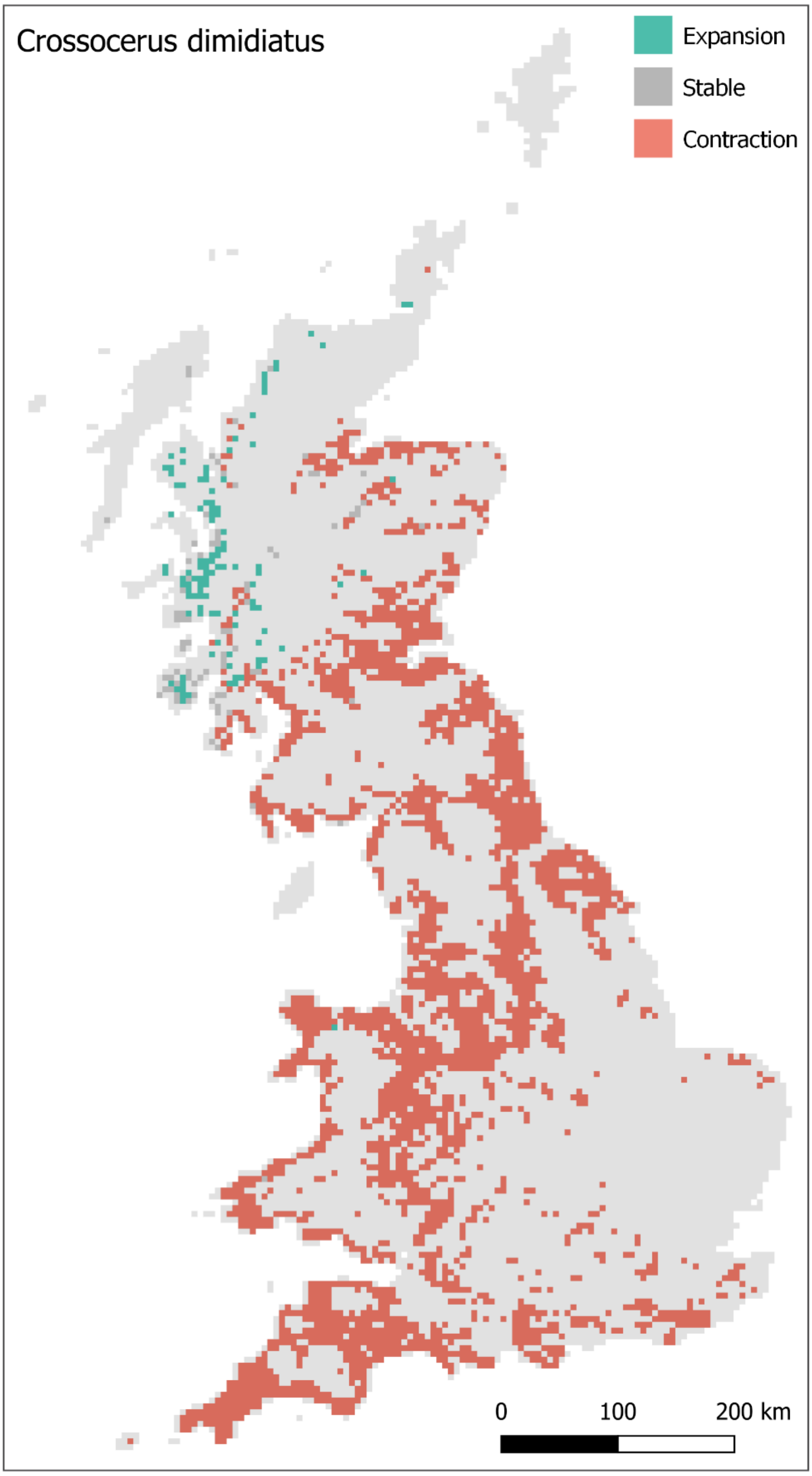

Figure C.12. Projected distribution change under climate change by 2040 of the Blunt Tailed Digger Wasp (Crossocerus dimidiatus).

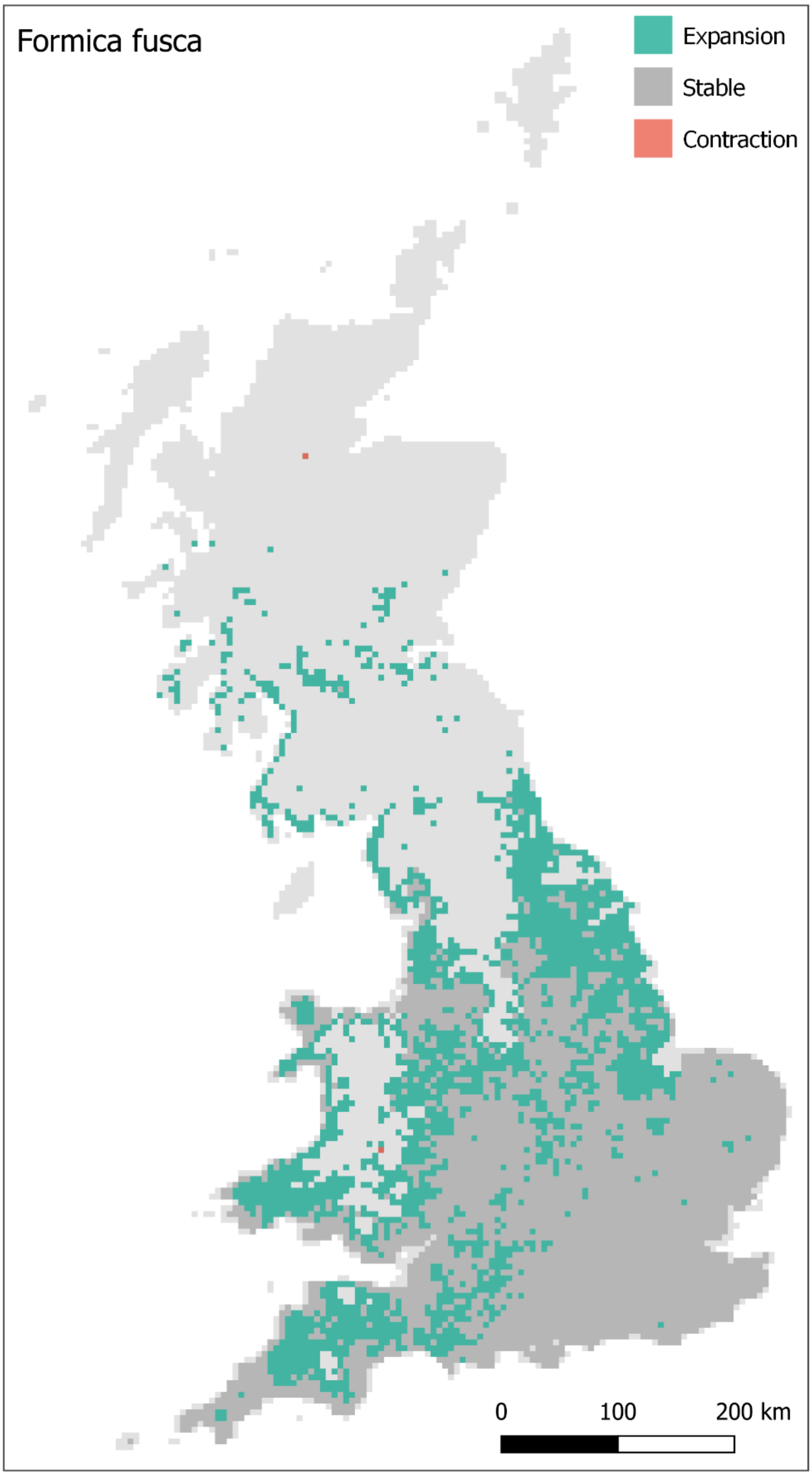

Figure C.13. Projected distribution change under climate change by 2040 of the Silky Ant (Formica fusca).

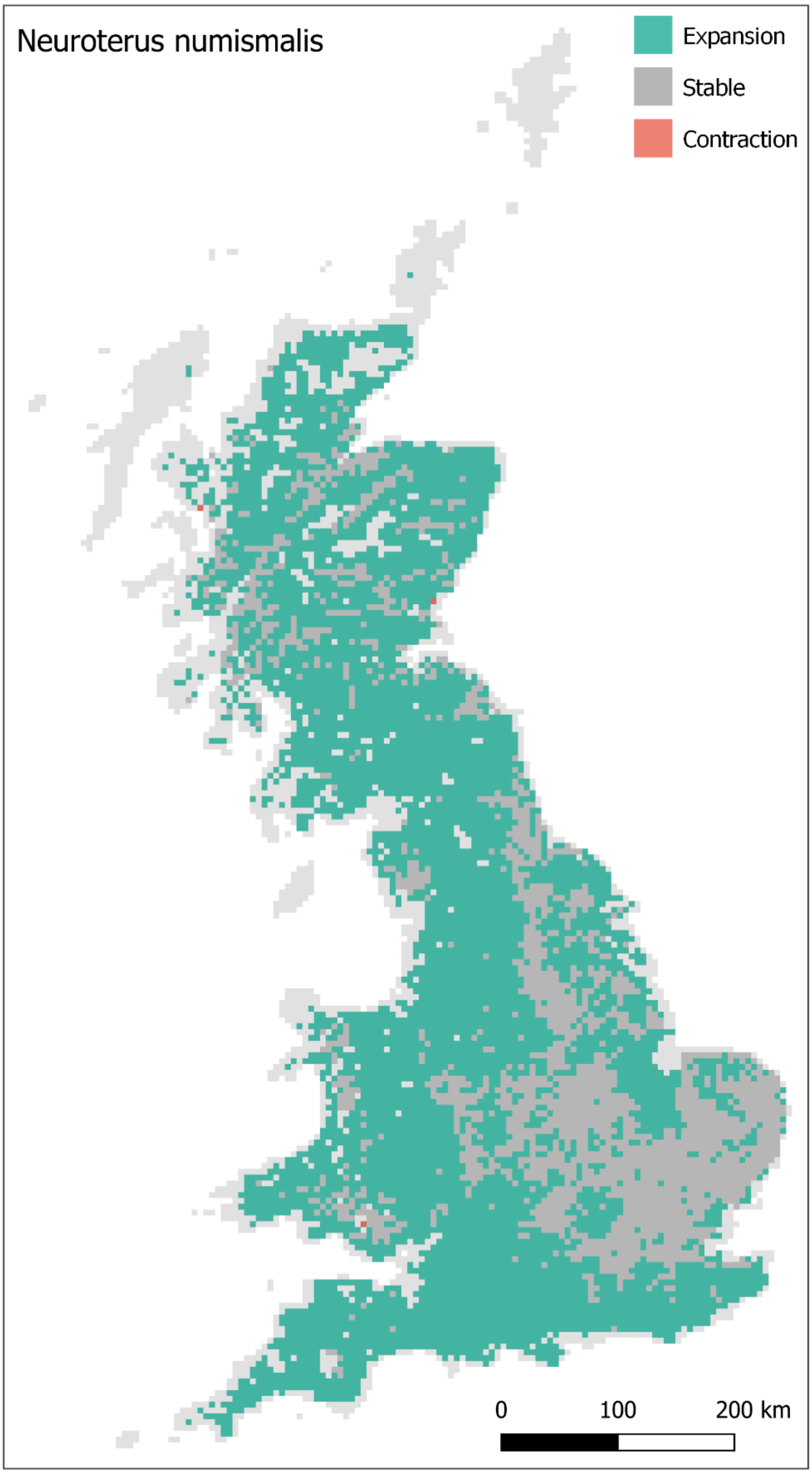

Figure C.14. Projected distribution change under climate change by 2040 of the Silk Button Gall Wasp (Neuroterus numismalis).

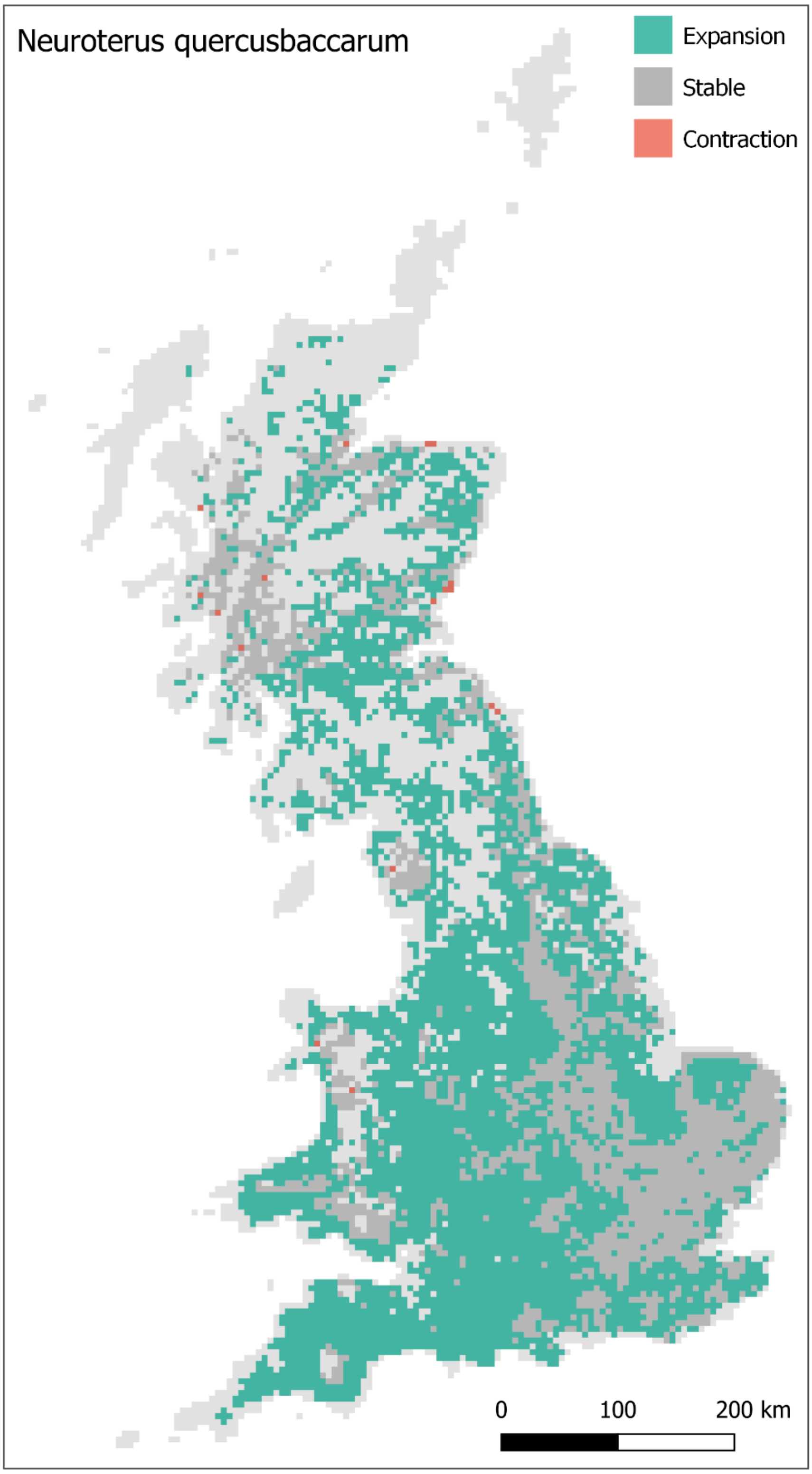

Figure C.15. Projected distribution change under climate change by 2040 of the Spangle Gall Wasp (Neuroterus quercusbaccarum).

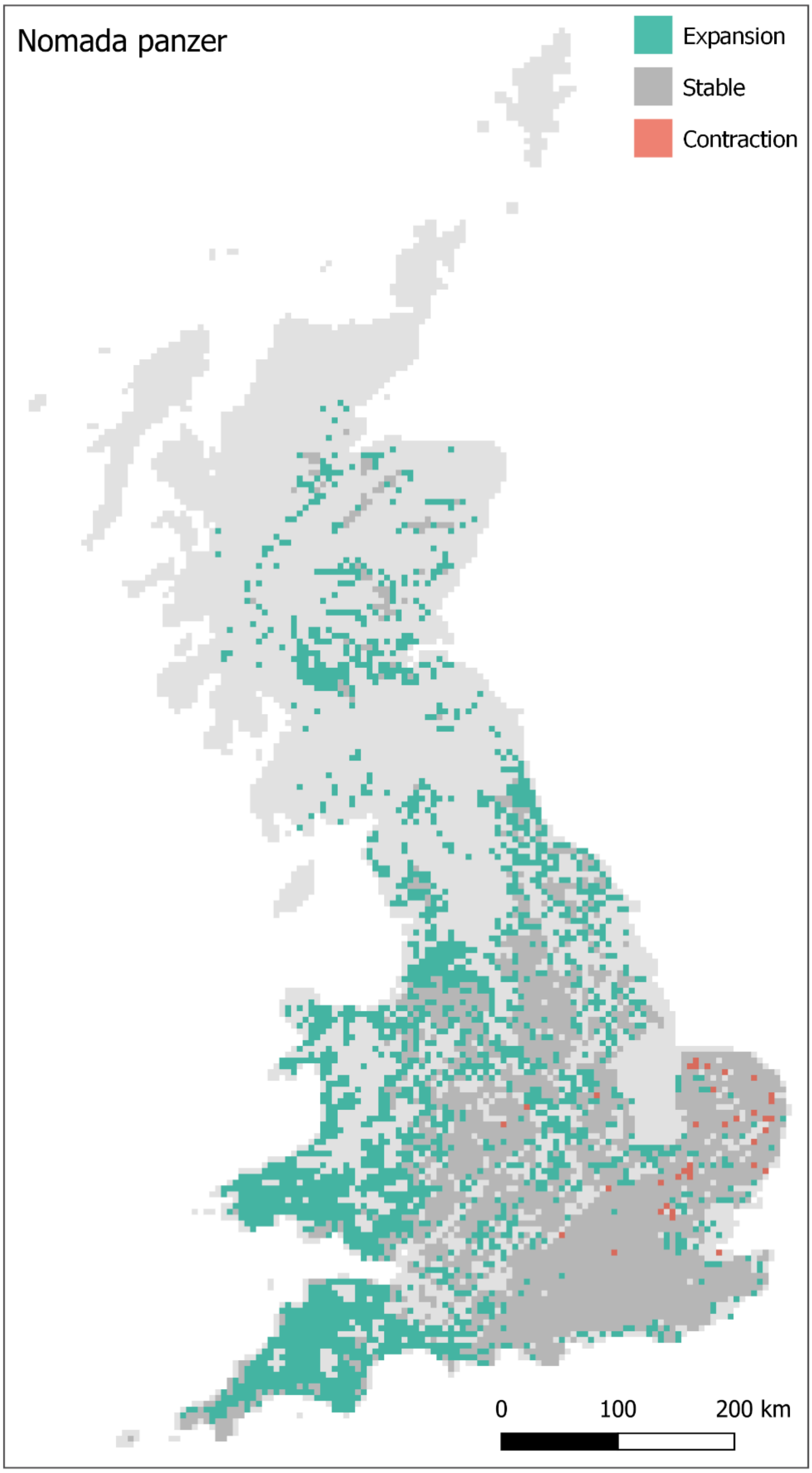

Figure C.16. Projected distribution change under climate change by 2040 of the Panzer's Nomad Bee (Nomada panzer).

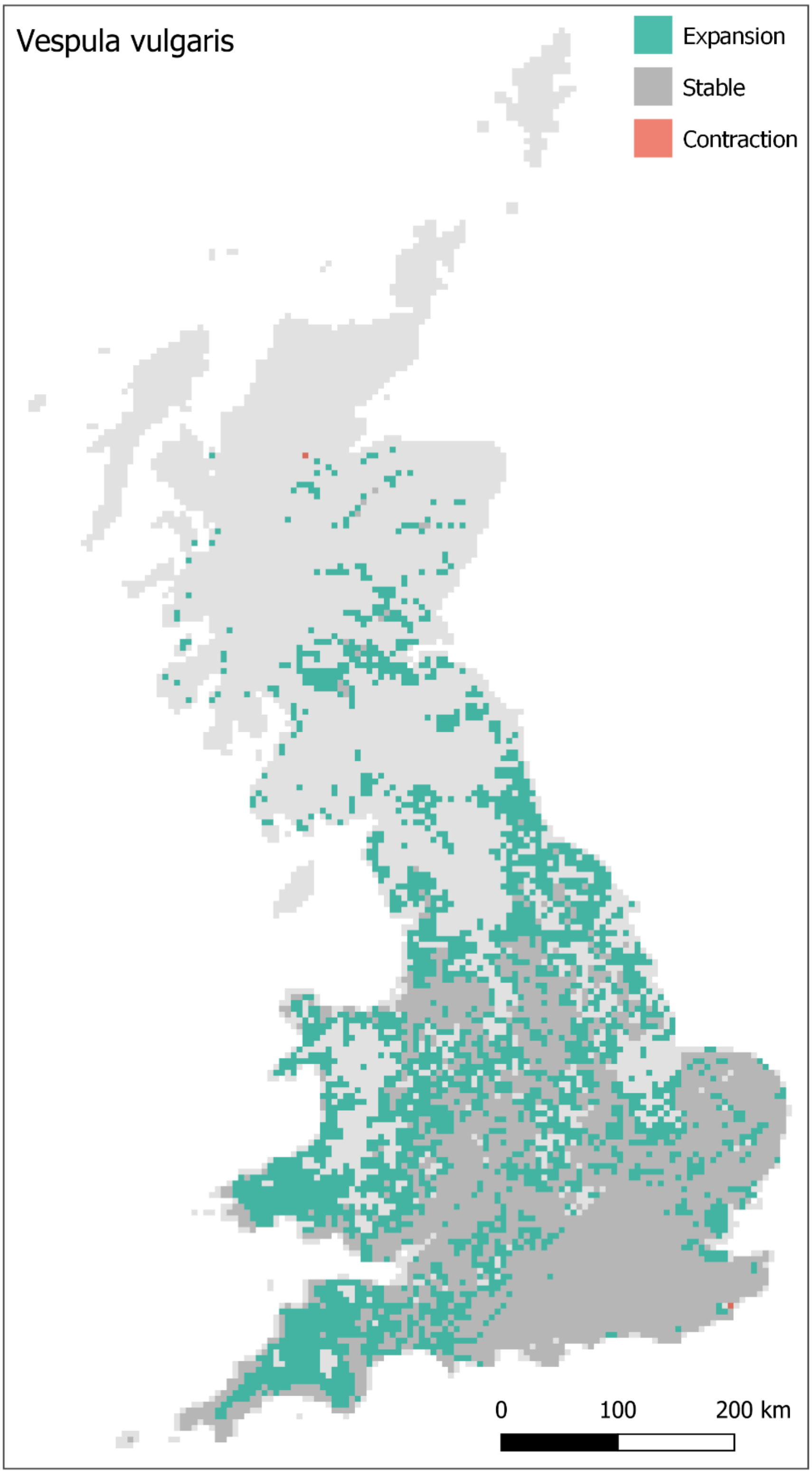

Figure C.17. Projected distribution change under climate change by 2040 of the European Wasp (Vespula vulgaris).

Appendix D: EICAT and EICAT+ results

Full results of the EICAT and EICAT+ literature search for impacts of Odonata and Hymenoptera study species on other UK native taxa. Each individual reported impact is included, detailing the impact mechanism, EICAT/EICAT+ impact category, evidence for the category chosen, sources of uncertainty and associated overall confidence rating with justification, as well as details of the impacted native species.

Table D.1. Full EICAT and EICAT+ search results. Sources of uncertainty are abbreviated as follows: CE – confounding effects, SD – study design, DQ – data quality, S – scale, C – coherence, E – extrapolation.

Species	Mechanism	Category	Evidence for impact category	Sources of uncertainty	Confidence rating and justification	Impacted native species	Reference
Andrena cineraria	Dispersal facilitation (through mutualism)	ML+	Nine species were recognised as effective pollen vectors including A. cineraria - insects leaving through the posterior openings, touching the androecium and demonstrated to carry pollen. For the long-term survival of populations, the protection of pollinator and their habitats is also essential.	CE:high SD:low DQ:medium S:low C:high E:low	Low: Extrapolation from Sweden; low pollination specificity of C. calceolus.	Cypripedium calceolus	1
	Dispersal facilitation (through mutualism)	ML+	The pollen diets of wild solitary bees were quantified using direct observations and pollen load analysis. Pollen grain for following species collected: Chaerophyllum temulum; Crataegus monogyna; Heracleum sphondylium; Medicago lupulina; Reseda lutea; Rhamnus cathartica; Rosa canina; Sisymbrium officinale. Visits to Brassica rapa campestris and Daucus carota for pollen also identified.	CE:high SD:low DQ:high S:medium C:medium	Low: Extensive surveys on farms in Hampshire and West Sussex between 2013 and 2015; assumptions.	Brassica rapa campestris; Chaerophyllum temulum; Crataegus monogyna; Daucus carota; Heracleum sphondylium; Medicago lupulina; Reseda lutea; Rhamnus cathartica; Rosa canina; Sisymbrium officinale	2

	Dispersal facilitation (through mutualism)	ML+	Visits to oilseed rape by A. cineraria observed along transects.	CE:medium SD:low DQ:medium S:low C:high	Low: Only visits recorded, pollination load not assessed.	Brassica napus	3
	Dispersal facilitation (through mutualism)	ML+	Mean body size of A. cineraria was related to the proximity of oilseed rape and abundance higher in grasslands surrounded by large areas covered by oilseed rape. Resulting higher abundance could exacerbate the competition for resources with other species.	CE:low SD:low DQ:medium S:low C:medium E:low	Low: A. cineraria is highly polylectic, foraging in a wide range of plant families, pollination not directly measured; extrapolation from Eastern France.	Brassica napus	4
	Provision of trophic resources (through parasitism)	ML+	Fifty newly emerged adults and 153 pupal exuviae of Bombylius major were counted during emergence at a colony of the bee Andrena cineraria. Parasitism of about 8% was estimated.	CE:high SD:low DQ:medium S:low C:high	Low: Local small scale study, error likely large and several assumptions made.	Bombylius major	5
	Provision of trophic resources (through parasitism)	ML+	Myopa testacea was present in three A. cineraria females.	CE:high SD:low DQ:medium S:low C:high E:low	Low: Parasitoids of other bees and wasps, here present in one Osmia cornuta female also; extrapolation from Belgium.	Myopa testacea	6
Andrena fulva	Dispersal facilitation (through mutualism)	ML+	28 visits (6.4% total visits) of A. Fulva to Anthyllis vulneraria recorded during 32 hours of field observations.	CE:medium SD:low DQ:low S:low C:medium E:low	Low: Only visits recorded, pollination load not assessed; extrapolation from Northwest Spain; low contribution compared to other species reported.	Anthyllis vulneraria subsp. vulgaris	7

	Dispersal facilitation (through mutualism)	ML+	The pollen diets of wild solitary bees were quantified using direct observations and pollen load analysis. Pollen grain for Acer pseudoplatanus was collected. Visits to Brassica rapa campestris for pollen also identified.	CE:medium SD:low DQ:medium S:medium C:medium	Low: Extensive surveys on farms in Hampshire and West Sussex between 2013 and 2015; assumptions.	Acer pseudoplatanus; Brassica rapa campestris	2
	Dispersal facilitation (through mutualism)	ML+	Visits to oilseed rape by A. fulva observed along transects.	CE:medium SD:low DQ:low S:low C:medium	Low: Only visits recorded, pollination load not assessed.	Brassica napus	3
	Provision of trophic resources (through predation)	ML+	Great titmice can periodically feed on Apoidea to a considerable degree, including Andrena fulva.	CE:medium SD:low DQ:medium S:low C:medium E:low	Low: Bees picked apart by great titmice were collected in the Botanical garden of Kiel; extrapolation from Germany.	Paris major	8
Andricus quercuscalicis	Indirect impacts through interactions with other taxa	ML+	49,406 galls in Britain were analysed and seven species were found to be parasitoids of the inquilines of A. quercuscalicis.	CE:high SD:low DQ:high S:high C:high	Low: Large scale study with thousands of galls investigated, however difficult to assess impacts for parasite species	Cecidostiba semifascia; Eupelmus urozonus; Eurytoma brunniventris; Mesopolobus amaenus; Mesopolobus jucundus; Torymus nitens	9
	Parasitism	MC-	Between 21% and 35% Quercus robur acorns were galled by A. quercuscalicis. The relative size of the acorn crop was not related to its rate of galling.	CE:high SD:medium DQ:high S:medium C:high	Medium: Assessment across several sites in south-east England; quantification of acorn production and percentage of acorns galled; difficult to determine population level impact.	Quercus robur	10

	Parasitism	MN-	A variable number of Quercus robur were attacked by the gall-forming wasp Andricus quercuscalicis (galls were present on up to 27% shoots). Insecticide treatment was effective in virtually eliminating gall-forming and sap-sucking insects and resulted in a significant increase in acorn production, however, this is not simply an artefact of excluding A. quercuscalicis.	CE:low SD:low DQ:medium S:low C:medium	Low: There is some indication of reduced performance due to A. quercuscalicis, however, results are not conclusive due to the influence of herbivorous insects also investigated.	Quercus robur	11
	Parasitism	MN-	A. quercuscalicis can have dramatic impact on the fecundity of Q. robur, attacking up to 90% of the entire acorn crop in some years, and consistently taking 100% of the acorns from certain individual trees. The consequence of this high level of seed mortality on the recruitment dynamics of the tree requires investigation, but there is no clear evidence that oak regeneration is acorn limited.	CE:high SD:low DQ:medium S:medium C:medium	Low: Parasitism observed over a long-term study (here results present first 10 years) of a site in England; further investigation required to determine population level impacts.	Quercus robur	12
	Provision of habitat (through commensalism)	ML+	49,406 galls in Britain were analysed revealing two species of inquilines (Synergus gallaepomiformis; S. umbraculus) that do not typically kill wasp larva.	CE:high SD:low DQ:high S:high C:high	Low: Large scale study with thousands of galls investigated, however difficult to assess impacts for parasite species	Synergus gallaepomiformis; Synergus umbraculus	9
	Provision of habitat (through commensalism)	ML+	The following inquilines emerged from the agamic galls: Synergus gallaepomiformis (n = 4959); Synergus nervous (n = 26); Synergus umbraculus (n = 2).	CE:high SD:low DQ:high S:medium C:medium	Low: Four study sites across Great Britain with incidences and rate of parasitism recorded; difficult to determine population level	Synergus gallaepomiformis; Synergus nervous; Synergus umbraculus	13

					impacts for parasitoids and inquiline species.		
	Provision of habitat (through commensalism)	ML+	The inquiline Aulogymnus gallarum was observed.	CE:high SD:low DQ:medium S:low C:medium	Low: Small scale study where 25 galls were studied; extrapolation from Romania.	Aulogymnus gallarum	14
	Provision of habitat (through commensalism)	MO+	Several inquiline species were identified from 38,901 knopper galls collected across 200 sites in Britain and Ireland. There has been a general upward trend in abundance of parasitoid and inquiline species over 15 year period, however, rates of parasitism remain low (< 6%).	CE:low SD:low DQ:medium S:medium C:medium	Low: Extensive study in Britian and Ireland, large sample size; difficult to determine population level impact from parasitism rates on single species.	Synergus gallaepomformis; Synergus umbraculus	15
	Provision of trophic resources (through parasitism)	ML+	Incidences of parasitoids and inquilines of Andricus quercuscalicis were collated from published literature. Rates of parasitism are generally low (less than 5% in 27 out of 31 cases, with a maximum rate of 16.6% recorded). Across 12 studies, parasitism of 131 individuals of Mesopolobus amaenus was reported, 29 Mesopolobus jucundus, 27 Gelis formicarius, 12 Mastrus castaneus, 3 Sycophila biguttata, 2 Eupelmus urozonus, and one individual of Arthrolytus ocellus, Spilomicrus stigmaticalis, Torymus cyaneus and Torymus geranii.	CE:high SD:low DQ:high S:medium C:medium	Low: Incidence of parasitism assessed across multiple UK based studies with rate of parasitism measured in several cases.	Arthrolytus ocellus; Eupelmus urozonus; Gelis formicarius; Mastrus castaneus; Mesopolobus amaenus; Mesopolobus jucundus; Spilomicrus stigmaticalis; Sycophila biguttata; Torymus cyaneus; Torymus geranii.	16

	Provision of trophic resources (through parasitism)	ML+	A. quercuscalicis is attacked by a number of generalist parasitoids, including Mesopolobus fuscipes, M. xanthocerus, M. tibialis and M. dubius, causing an overall loss of around 25% of all sexual galls each year.	CE:high SD:low DQ:high S:low C:medium	Low: Sampling of parasitism of 1000 galls a year for an eight year period.	Mesopolobus dubius; Mesopolobus fuscipes; Mesopolobus tibialis; Mesopolobus xanthocerus	12
	Provision of trophic resources (through parasitism)	ML+	Parasitism by Mesopolobus dubius, Mesopolobus fuscipes, Mesopolobus tibialis, and Mesopolobus xanthocerus caused mortality of 455 galls in 1986 and 890 in 1989.	CE:high SD:low DQ:medium S:low C:medium	Low: Investigation of gall mortality on 5 Quercus cerris trees in Silwood Park, Berkshire.	Mesopolobus dubius; Mesopolobus fuscipes; Mesopolobus tibialis; Mesopolobus xanthocerus	17
	Provision of trophic resources (through parasitism)	ML+	49,406 galls in Britain were analysed and four parasitoids of A. quercuscalicis observed (Gelis formicarius; Ormyrus nitidulus; Megastigmus stigmatizans; Sycophila biguttata).	CE:high SD:low DQ:high S:high C:high	Low: Large scale study with thousands of galls investigated, however difficult to assess impacts for parasite species	Gelis formicarius; Ormyrus nitidulus; Megastigmus stigmatizans; Sycophila biguttata	9
	Provision of trophic resources (through parasitism)	ML+	Several parasitoid species (Aprostocetus aethiops; Mesopolobus dubius; Mesopolobus fuscipes; Mesopolobus tibialis; Mesopolobus xanthocerus) were observed from a collection of 1000 catkins over five trees at the study site in Oxford UK. Parasitoids caused a combined mortality of 21.7% for A. quercuscalicis	CE:high SD:low DQ:medium S:low C:high	Low: Samples from 5 trees at one study site; little indication of impact for parasite species	Aprostocetus aethiops; Mesopolobus dubius; Mesopolobus fuscipes; Mesopolobus tibialis; Mesopolobus xanthocerus	18
	Provision of trophic resources (through parasitism)	ML+	Dissection of 1050 galls revealed parasitism by Ormyrus nitidulus at a rate of 20%.	CE:high SD:low DQ:high S:medium C:high	Low: 1050 galls dissected from a range of localities within Britain.	Ormyrus nitidulus	19

	Provision of trophic resources (through parasitism)	ML+	Parasitoid attack rates in the sexual galls of A. Quercuscalicis was 0.01 for Mesopolobus dubius across UK sites, 0.06 to 0.3 for Mesopolobus fuscipes, 0.01 to 0.18 for Mesopolobus tibialis and 0.03 to 0.12 for Mesopolobus xanthocerus. The following parasitoids emerged from the agamic galls: Cecidostiba hilaris (n = 931); Eupelmus urozonus (n = 109); Eurytoma brunniventris (n = 429); Mesopolobus amaenus (n = 9); Mesopolobus sericeus (= jucundus; n = 979); Mesopolobus tibialis (n = 4); Ormyrus nitidulus (n = 8); Sycophila biguttara (n = 37); Torymus geranii (n = 6); Torymus nitens (n = 87).	CE:high SD:low DQ:high S:medium C:medium	Low: Four study sites across Great Britain with incidences and rate of parasitism recorded; difficult to determine population level impacts for parasitoids and inquiline species.	Cecidostiba hilaris; Eupelmus urozonus; Eurytoma brunniventris; Mesopolobus amaenus; Mesopolobus dubius; Mesopolubus fuscipes; Mesopolobus sericeus; Mesopolobus tibialis; Mesopolobus xanthocerus; Ormyrus nitidulus; Sycophila biguttara; Torymus geranii; Torymus nitens	13
	Provision of trophic resources (through parasitism)	ML+	The parasite Mesopolobus tibialis was identified for Andricus quercuscalicis.	CE:high SD:low DQ:medium S:low C:medium	Low: Small scale study where 25 galls were studied; extrapolation from Romania.	Mesopolobus tibialis	20
	Provision of trophic resources (through parasitism)	MO+	Several parasitoid species were identified from 38,901 knopper galls collected across 200 sites in Britain and Ireland. There has been a general upward trend in abundance of parasitoid species over 15 year period, however, rates of parasitism remain low (< 6%).	CE:low SD:low DQ:medium S:medium C:medium	Low: Extensive study in Britian and Ireland, large sample size; difficult to determine population level impact from parasitism rates on single species.	Cecidostiba semifascia; Eurytoma brunniventris; Gelis formicarius; Megastigmus stigmatizans; Mesopolobus jucundus; Mesopolobus amaenus; Ormyrus nitidulus; Sycophila biguttara; Torymus nitens	15

	Provision of trophic resources (through predation)	ML+	Galls of Q. cerris suffered from predation by small insectivorous birds, mainly blue tits, Parus caerulea (Cyanistes caeruleus).	CE:medium SD:low DQ:medium S:low C:medium	Low: Predation observed but rate and impact not assessed.	Cyanistes caeruleus	12
Bombus soroeensis	Dispersal facilitation (through mutualism)	ML+	2 visits (0.001% total visits) of Bombus soroeensis to Vaccinium uliginosum recorded during 298 hours of field observations.	CE:high SD:low DQ:high S:low C:low E:low	Low: Only visits recorded, pollination load not assessed; extrapolation from Belgium; low contribution compared to other species reported.	Vaccinium uliginosum	21
Cerceris arenaria	Predation	MC-	Several prey species of C. arenaria were identified (3842 specimens collected in total during the 6 year study), with the most prevalent (> 10%) across the study sites being Sitona hispidulus, Hypera nigrirostris, Otiorhynchus rugosostriatus and Otiorhynchus armadillo.	CE:high SD:low DQ:high S:medium C:medium E:low	Low: Prey specimens collected across 8 years of study; extrapolation from two study sites in Northern Italy.	Cosmobaris scolopacea; Hypera postica; Hypera rumicis; Hypera nigrirostris; Hypera pollux; Hypera zoilus; Lepyrus capucinus; Otiorhynchus armadillo; Otiorhynchus crataegi; Otiorhynchus ovatus; Otiorhynchus rugosostriatus; Otiorhynchus sulcatus; Polydrusus formosus; Sitona hispidulus; Sitona humeralis; Sitona sulcifrons	22
	Predation	MO-	Predation of several weevil species by Cerceris arenaria was observed. The average number of prey brought to a nest, calculated for the whole period of the study (except rainy days), was of 2.05 per day per wasp (range =	CE:medium SD:medium DQ:medium S:low E:low	Low: Prey reported and nests dissected; extrapolation from study site in Northern Italy.	Hypera postica; Hypera rumicis; Hypera zoilus; Lepyrus capucinus; Otiorhynchus ovatus; Otiorhynchus rugosostriatus; Otiorhynchus	23

			0–4.53; SD = 1.08; n = 34). At this rate approximately 40,000 weevils could be destroyed in the study area during the whole season (average 2.05 prey per day for average 20.73 days across 907 recorded nests) having implications in terms of biological pest control.			sulcatus; Polydrusus formosus; Sitona lepidus; Sitona hispidulus; Sitona sulcifrons ssp. Argutulus	
	Provision of trophic resources (through parasitism)	ML+	Hedychrum nobile was observed during the summer of 2005 at a large aggregation (about 300 nests) of the digger wasp C. arenaria. A total 119 brood cells were found during excavations at the study site. The overall parasitism rate was 9.2% with a mortality rate of C. arenaria of 25.2%	CE:high SD:low DQ:high S:low C:high E:low	Low: Extrapolation from single study site in Northern Italy.	Hedychrum nobile	24
	Provision of trophic resources (through parasitism)	MR+	Population trends of cuckoo wasps (Chrysidiae) and hosts species were positively correlated with C. arenaria and H. nobile experiencing increases in occupancy of 73.3% and 32.7%, respectively.	CE:medium SD:medium DQ:medium S:low C:medium E:low	Low: Based on significant relationship across selection of kleptoparasitic and parasitoid species and their hosts; extrapolation from Finland; potential sources of bias from change in sampling methods and accuracy of collection localities during study period.	Hedychrum nobile	25
Crossocerus	Predation	MC-	Eight C. dimidiatus nests were discovered in Danby, North Yorkshire Moors. Nest excavations revealed	CE:high SD:low DQ:medium S:low C:medium	Low: Nests excavated and prey reported; small scale local study.	Hylemyia variata; Hylemyia strenua	26

			prey species including 8 individuals of Hylemyia variata (Hylemya variata) and 7 Hylemyia strenua (Hylemya stenua).	E:low			
Formica fusca	Dispersal facilitation (through mutualism)	MN+	Ant species, including Formica fusca, provided dispersal services to the myrmecochorous plant Knautia arvensis, with ant dispersal resulting in a more even and on average longer distances in comparison to unassisted dispersal. The maximum distance of a dispersal event was 994cm by a formica fusca individual. For formica fusca there were 4 observations of individuals carrying K. arvensis seeds, consisting of 13.3% dispersal events recorded during the day.	CE:low SD:medium DQ:medium S:low C:medium E:low	Low: Although adapted for ant dispersal and the reported dispersal services, the impact of formica fusca specifically for K. arvensis is unclear; extrapolation from the Czeck Republic.	Knautia arvensis	27
	Indirect impacts through interactions with other taxa	ML+	A colony of Myrmica sabuleti ants profited from a Polygerus rufescens raid against a Formica fusca nest, by collecting carcasses to be used as food. 27 carcasses were collected, constituting approximately half of the dry biomass weight of all M. sabuleti workers.	CE:medium SD:low DQ:low S:low C:low E:low	Low: Although the amount of food gained by M. sabuleti was significant, it is difficult to assess the potential indirect impact of slave makers' raids on F. fusca, particularly as the supply of this food to non-host ants is likely occasional only; extrapolation from the Czech Republic.	Myrmica sabuleti	28
	Indirect impacts through interactions with other taxa	MN-	Pollen exposed to Formica fusca was inhibited by an average of approximately 20%. Exposure to F. fusca	CE:high SD:low DQ:medium S:low C:low	Low: Small scale study where pollen germination impact by	Teucrium scorodonia; Valeriana officinalis	29

			caused a significant reduction in the pollen germination rate of Valeriana officinalis and Teucrium scorodonia.		Formica fusca was investigated for a range of British plants, however, population level impacts not investigated.		
	Predation	MC-	Of ten replicas, all Acrolepiopsis assectella caterpillars were attacked by Formica Fusca predators, with one being highly eaten and four being partially eaten.	CE:high SD:low DQ:low S:low C:medium E:low	Low: Predation events observed but impact on performance and population unassessed; small scale study; extrapolation from France.	Acrolepiopsis assectella	30
	Provision of trophic resources (through parasitism)	ML+	Both nests of Formica sanguinea investigated where populated by F. Fusca workers as slaves.	CE:low SD:low DQ:low S:low C:medium E:low	Low: Small scale study where only two nests were investigated; whilst F. fusca provided a function through social parasitism the study did not allow for investigation of the impacts; extrapolation from Italy.	Formica sanguinea	31
	Provision of trophic resources (through parasitism)	ML+	Social parasitism of Formica fusca was observed in colonies of Formica polyctena and Polygerus rufescens.	CE:low SD:low DQ:low S:low C:medium E:low	Low: No investigation of impacts for slave-making colonies; extrapolation from Poland.	Formica polyctena; Polygerus rufescens	32
	Provision of trophic resources (through parasitism)	ML+	Social parasitism of Formica fusca was observed in colonies of Formica sanguinea and Polygerus rufescens.	CE:low SD:low DQ:low S:low C:medium E:low	Low: No investigation of impacts for slave-making colonies; extrapolation from Poland.	Formica sanguinea; Polygerus rufescens	33
	Provision of trophic resources (through parasitism)	ML+	Two colonies of Polygerus rufescens were observed with Formica fusca slaves.	CE:low SD:low DQ:low S:low C:medium E:low	Low: No investigation of impacts for slave-making colonies; extrapolation from Poland.	Polygerus rufescens	34

	Provision of trophic resources (through parasitism)	ML+	11 Formica sanguinea colonies were sampled, Formica fusca slaves were present constituting between 9 to 54% (mean 34%) proportion of individuals in the colonies.	CE:low SD:low DQ:low S:low C:medium E:low	Low: Although constituting up to 54% in F. sanguinea colonies, the subsequent impact level has not been reviewed; extrapolation from Poland.	Formica sanguinea	35
	Provision of trophic resources (through mutualism)	MN+	The presence and number of Formica fusca ants had a significant positive effect on the fitness of fireweed aphids (Aphis varians), showing ants are a limited and limiting resource for a tended fireweed aphid species.	CE:high SD:low DQ:medium S:low C:low	Low: Experimental study over two years revealing that ant species such as Formica fusca are a limiting resource for tended Aphis varians with fitness and performance being improved by these species, however, population level impacts are not investigated.	Aphis varians	36
	Provision of trophic resources (through mutualism)	MO+	Formica fusca regulated populations of leafhopper (Dalbulus quinquenotatus) by controlling the size and length of development of leafhopper populations. By keeping leafhopper populations smaller, tending ants were able to ensure the continued survival of the colony by avoiding overpopulation of the host plant and subsequent extirpation of the colonies. Ants were also observed protecting leafhoppers from nabid predators.	CE:medium SD:medium DQ:medium S:low C:medium E:low	Low: The population dynamics and impacts of tending Formica fusca ants were tested in the laboratory on a small scale, furthermore the host plant Tripsacum dactyloides and predator Nabis americoferus employed in the experiment are not native to the UK and so it is unclear whether UK impacts would be similar;	Dalbulus quinquenotatus	37

					extrapolation from US and Mexico.		
	Provision of trophic resources (through predation)	ML+	Gastropoda, Arachnida and insects represented 97% Tetrao urogallus food by wet weight, with Formica fusca being one of the predominant insects included.	CE:high SD:low DQ:low S:low C:medium E:low	Low: Diet of species was investigated however the importance of the Formica fusca species is difficult to ascertain; extrapolation form Russia.	Tetrao urogallus	38
	Provision of trophic resources (through predation)	ML+	Analysis of Jynx torquilla nestling faecal sacks revealed that Formica fusca was an important food source, though not as important as Lasius Niger and Tetramorium caespitum. Formica fusca constituted approximately 13% ant colonies and contributed the same proportion to the diet of Jynx torquilla on average.	CE:high SD:low DQ:medium S:medium C:medium E:low	Low: Although providing an indication of the importance of Formica fusca to the diet of the Northern wryneck nestling's surveyed, it is difficult to determine the subsequent impact level; extrapolation form Denmark.	Jynx torquilla	39
Neuroterus numismalis	Parasitism	MC-	Random samples of 200 leaves were inspected for galls, Neuroterus numismalis was routinely encountered.	CE:high SD:low DQ:medium S:low C:high	Low: Small scale study of 36 trees. Galls encountered, impact not studied.	Quercus robur	40
	Parasitism	MC-	Field studies were conducted at three localities in Southern Poland, at one site galls of Neuroterus numismalis were present on 12 out of 20 trees (60%), on 47 out of 2000 leaves (0.023%), with 67 galls recorded in total on analysed	CE:high SD:low DQ:high S:low C:low E:low	Low: Extrapolation from observations in Poland; low parasitism by Neuroterus numismalis compared to other species; variation in	Quercus robur	41

			leaves. At another site a single gall was observed.		parasitism rate betwen study sites.		
	Parasitism	MC-	Galls of Neuroterus numismalis were observed on Quercus petraea.	CE:high SD:low DQ:medium S:low C:medium E:low	Low: Single observation; extrapolation from Turkey.	Quercus petraea	42
	Parasitism	MN-	The leaves with galls of N. numismalis were characterized by an extremely high decrease of Chlorophyll a and b contents, reduced by 61.39% and 65.65%, respectively, when compared to non-infested leaves. Photosynthetic parameters were significantly decreased. This indicates that infestations had a significant impact on physiology, biochemistry, morphology and anatomy of oak leaves.	CE:high SD:low DQ:high S:low C:medium E:low	Low: Comprehensive investigation of the impacts of N. numismalis galls on Q. robur, further investigation required to determine population level impact of altered performance caused by N. numismalis; extrapolation from Poland	Quercus robur	43
	Provision of habitat (through commensalism)	ML+	Inquiline Synergus incrassatus emerged from gall of N. numismalis.	CE:high SD:low DQ:medium S:low C:medium E:low	Low: Single observation; extrapolation from Turkey.	Synergus incrassatus	42
Neuroterus quercusbaccarum	Parasitism	MC-	Random samples of 200 leaves were inspected for galls, Neuroterus quercusbaccarum was routinely encountered and was most abundant of the cynipid species encountered.	CE:high SD:low DQ:high S:low C:medium	Low: Small scale study of 36 trees. Galls encountered, impact not studied.	Quercus robur	40
	Parasitism	MC-	6730 galls of N. quercusbaccarum were found across eight oak species, including Britain's native oak species.	CE:high SD:low DQ:high S:low C:medium E:low	Low: Small scale study; galls encountered, impact not studied; extrapolation	Quercus petraea; Quercus robur	44

					from Southern Slovakia.		
	Parasitism	MC-	Field studies were conducted at three localities in Southern Poland, galls of Neuroterus quercusbaccarum were present on between 8 (40%) and 17 (85%) trees, depending on the site. A total of 756 galls were recorded on analysed leaves.	CE:high SD:low DQ:high S:low C:medium E:low	Low: Extrapolation from observations in Poland; variation in parasitism rate betwen study sites.	Quercus robur	41
	Parasitism	MC-	2786 spangle galls of N. quercusbaccarum were counted on 120 branches (6000 leaves) of oak, with a mean infestation rate of 0.29 galls per leaf, excluding outliers.	CE:high SD:low DQ:high S:low C:medium E:low	Low: Small scale study; galls encountered, impact not studied; extrapolation from Switzerland.	Quercus petraea; Quercus robur	45
	Parasitism	MN-	The leaves with galls of N. quercusbaccarum were characterized by an decrease of Chlorophyll a and b contents as well as decreased photosynthetic parameters, when compared to non-infested leaves.	CE:high SD:low DQ:high S:low C:high E:low	Low: Comprehensive investigation of the impacts of N. Quercusbaccarum galls on Q. robur, further investigation required to determine population level impact of altered performance; extrapolation from Poland.	Quercus robur	43
	Parasitism	MN-	Data generated in this study indicate that the development of N. quercusbaccarum galls on pedunculate oak leaves has a negative effect on host plant related to the disruption of cell membrane integrity, disturbance of photosynthesis and	CE:high SD:low DQ:high S:low C:high E:low	Low: Comprehensive investigation of the impacts of N. Quercusbaccarum galls on Q. robur, further investigation required to determine population level impact of altered	Quercus robur	46

			reduction of the antioxidant potential of the host plant.		performance; extrapolation from Poland.		
	Provision of habitat (through commensalism)	ML+	The inquiline species Aulogymnus arsames and Torymus auratus emerged from the 667 galls collected.	CE:high SD:low DQ:high S:low C:medium E:low	Low: Parasites observed from galls; extrapolation from France.	Aulogymnus arsames; Torymus auratus	47
	Provision of habitat (through commensalism)	ML+	Inquiline Synergus tibialis emerged from gall of N. numismalis.	CE:high SD:low DQ:medium S:low C:medium E:low	Low: Single observation; extrapolation from Turkey.	Synergus tibialis	42
	Provision of trophic resources (through parasitism)	ML+	667 galls were collected and the parasitoid species Mesopolobus tibialis regularly emerged. Eurytoma brunniventris and Mesopolobus fasciiventris were also present in two galls and one gall, respectively.	CE:high SD:low DQ:high S:low C:medium E:low	Low: Parasites observed from galls; extrapolation from France.	Eurytoma brunniventris; Mesopolobus fasciiventris; Mesopolobus tibialis	47
Nomada panzeri	Dispersal facilitation (through mutualism)	MR+	Nomada panzeri identified as pollinator of lady's-slipper orchid. Results indicated significant loss (30%-63%) of suitable habitat of C. calceolus in 2070, but the pollination availability should not further limit the chance of survival of this species. Nomada panzeri, the only representative of Nomadinae, will occur in 25.30–52.96% of the predicted range of C. calceolus, depending on the scenario considered. Under rcp8.5, Nomada panzeri has the second most pollinator potential (after Syrphus ribesii).	CE:low SD:medium DQ:medium S:high C:medium	Low: Pollination potential inferred by projection of lady's-slipper and pollinator species distribution under climate change scenarios, there are other factors that can increase the extinction rate of the lady's-slipper orchid.	Cypripedium calceolus	48

Vespula vulgaris	Competition	MC-	A strong interference competition between Vespula vulgaris wasps and Apis mellifera was observed during laboratory experiments, with several cases of aggression noted, particularly when food sources were low. Field results were less conclusive, with some evidence of exploitation competition, however, V. vulgaris and A. mellifera largely had space enough to feed without encountering one another.	CE:low SD:low DQ:low S:low C:low E:low	Low: Conclusions can only be drawn with low confidence for several reasons. This study took place in New Zealand where V. vulgaris has been introduced and competition mainly reported with respect to honeydew food resources, produced by Ultraceolostoma brittini. It might therefore be inappropriate to extrapolate to the UK where both species are native and compete over different food sources. In addition, the results of this study are inconclusive as field observations were not coherent with laboratory findings.	Apis mellifera	49
	Competition	MC-	An improved competitive ability by V. vulgaris in comparison to V. germanica is suggested as V. germanica was found to avoid baits with visual and olfactory cues indicating the presence of V. vulgaris, whereas, V. vulgaris was not deterred by the presence of V. germanica.	CE:medium SD:low DQ:medium S:low C:low E:low	Low: The competitive advantage of V. vespula over V. germanica is largely speculatory, with an absence of direct evidence for competitive interactions; extrapolation from Argentina where both species are invasive.	Vespula germanica	50

	Dispersal facilitation (through mutualism)	ML+	Scrophularia umbrosa, is a wasp-flower mainly pollinated by social wasps. Both floral odour and visual cues were found to contribute to the pollinator attraction of V. vulgaris.	CE:medium SD:low DQ:medium S:low C:medium E:low	Low: Vespula vulgaris was attractive to the wasp-flower Scrophularia umbrosa, however, as this species has a generalised pollination system and is able to attract a broad spectrum of pollinators the impact of V. vulgaris is likely to be minimal; extrapolation from Germany.	Scrophularia umbrosa	51
	Dispersal facilitation (through mutualism)	ML+	Surveys showed that at 13% V. vulgaris was one of the main insect taxa foraging on ivy (Hedera helix and Hedera hibernica), however, behind hoverflies, other fliers and honey bees. The pollination Potential (PP) index score across wasps was calculated as 0.30 (with 0.35 for honey bees, 0.21 for hoverflies and 0.10 for bumblebees).	CE:medium SD:low DQ:low S:low C:low	Low: V. vespula was a frequent visitor to ivy, however, pollination impact was not measured. Although previously suggested that ivy is functionally specialised for pollination by Vespula species, this study indicated honey bees, bumble bees and hover flies are also important pollinators of ivy, therefore, the impact of V. vespula is likely minimal.	Hedera helix; Hedera hibernica	52
	Dispersal facilitation (through mutualism)	MN+	A highly specific and effective pollination system by Epipactis helleborine and E. purpurata was observed; by omitting green leaf volatiles that	CE:high SD:low DQ:medium S:low C:high E:low	Low: A highly specific pollination system was discovered for these species, therefore it is	Epipactis helleborine; Epipactis purpurata	53

			are usually produced by wounded plants infested by herbivores, these orchids are able to chemically mimic the presence of prey attractive to foragers of the social wasps V. vulgaris and V. germanica.		likely that V. vespula has a significant role in pollination, particularly in areas where V. germanica is absent, however, further research is required to determine the impact level; extrapolation from Austria.		
	Predation	MC-	A predation event of Anthocharis cardamines caterpillars by Vespula vulgaris was reported.	CE:low SD:low DQ:low S:low C:low E:high	Low: Observation of predation reported but impact level not assessed.	Anthocharis cardamines	54
	Predation	MO-	Predation of V. vulgaris was found to have a high impact on Pieris rapae caterpillars and Athalia rosae larvae. P rapae, in particular were killed within hours by predatory wasps with authors determining that it is likely that places where activity of V. vulgaris is high, P. rapae has no chance to survive.	CE:low SD:low DQ:medium S:low C:medium E:low	Low: Although predation by V. vespula appears significant, it is difficult to assess population level impacts with any confidence; extrapolation from the Netherlands.	Athalia rosae; Pieris rapae	55
	Provision of trophic resources (through parasitism)	ML+	Metoecus paradoxes is predominantly found in the nests of Vespula vulgaris, with the number of nests infested by the parasitoid varying from 20 to 67%. The number of beetles per nest typically varied between 1 and 25, with only a few cases where numbers exceeded 50.	CE:high SD:low DQ:medium S:medium C:high E:low	Low: Assessment of typical hosts of Metoecus paradoxus and parasitism rate within literature, it is difficult to ascertain impact level despite V. vulgaris being the most common host species; extrapolation from the Netherlands.	Metoecus paradoxus	56

	Provision of trophic resources (through parasitism)	ML+	Nest usurpation of V. vulgaris by V. germanica has been observed, a temporary form of social parasitism.	CE:low SD:low DQ:low S:low C:low E:low	Low: Report of social parasitism, however, limited evidence and no assessment of impact level; extrapolation from the United States.	Vespula germanica	57
	Provision of trophic resources (through parasitism)	ML+	Between 3.4% and 5.3% V. vespula nests were infested by the parasitoid Sphecophaga vesparum vesparum for the first 4 years after the parasitoid was released. After 1991, this was more variable, but consistently between 7.5% and 17.3% throughout the period 1992-1999. The level of parasitism did not increase significantly with time.	CE:low SD:low DQ:medium S:low C:low E:low	Low: Parasitism rate was variable across the study site; extrapolation from New Zealand.	Sphecophaga vesparum vesparum	58
	Provision of trophic resources (through parasitism)	ML+	Metoecus paradoxus almost exclusively parasitises nests of Vespula vulgaris and has been found to be chemically adapted to their main host, sharing more hydrocarbon compounds with it than they do the related V. germanica. Aggression tests also revealed that adult beetles were attached less by V. vespula than V. germanica.	CE:high SD:low DQ:medium S:low C:medium E:low	Low: The parasitoid beetle Metoecus paradoxus appears highly specialised for parasitism of V. vespula nests, however, specific impact levels were not reviewed; extrapolation from Belgium.	Metoecus paradoxus	59
	Provision of trophic resources (through parasitism)	ML+	Vespula vulgaris nests were investigated over three years for the presence of arthropod parasites. Volucella inanis was present in 40 out of a total 45 nests (88.9%), Volucella zonaria in 21 nests (46.7%), Metoecus	CE:high SD:low DQ:high S:medium C:medium	Low: Several arthropod parasites where present in Vespula vulgaris nests across England, however, resulting impact level requires	Metoecus paradoxus; Sphecophaga vesparum; Volucella inanis; Volucella zonaria	60

			paradoxus in 17 nests (37.8%) and Sphecophaga vesparum in 7 nests (15.6%).		further investigation.		
	Provision of trophic resources (through parasitism)	MO+	Thirteen generations of Sphecophaga vesparum vesparum were raised and all stages released into Vespula vesparum nests as biological control agents. Six of seven nests were parasitised with parasitoid numbers increasing approximately eight-fold.	CE:high SD:medium DQ:medium S:low C:medium E:low	Low: Small scale study, which did not investigate long term impact on parasitoid; extrapolation from New Zealand.	Sphecophaga vesparum vesparum	61
	Provision of trophic resources (through parasitism)	MO+	The European wasp parasitoid, sphecophaga vesparum vesparum was released at one location in New Zealand. Subsequent nest searches of Vespula vulgaris in the surrounding area indicated rapidly multiplying parasitoid population in two nests, 33km and 625km away from original release boxes.	CE:low SD:medium DQ:medium S:low C:medium E:low	Low: Evidence shows establishment and spread of parasitoid of Vespula vulgaris, however, extrapolation from New Zealand where the species has been introduced means the conclusions for the UK can only be made with low confidence.	Sphecophaga vesparum vesparum	62
	Transmission of disease	MC-	68 novel and 9 previously identified viruses sequences were found in transcriptomes of Vespula vulgaris. Experimental infection of honey bees (Apis mellifera) with one novel Moku-like virus resulting in an active infection.	CE:low SD:low DQ:medium S:low C:low E:low	Low: This virus was found to be infectious to honey bees, and there is high risk of viral spillover from V. vespula, with spillover events being widespread amongst Hymenoptera, however this effect requires further exploration;	Apis mellifera	63

					extrapolation from Belgium and New Zealand native and invasive specimens.		
Aeshna cyanea	Predation	MC-	Investigation into the effects of predator confusion on functional responses with predator Aeshna cyanea and Daphnia magna. The maximum number of prey eaten in 2 minutes was 8.6 (at a prey density 100).	CE:high SD:low DQ:high S:low C:medium E:low	Low: Laboratory experiment into the impact of swarming of Daphnia on A. cyanea; extrapolation 7from Germany.	Daphnia magna	64
	Predation	MC-	Laboratory experiment on whether Rana temporaria tadpoles account for density when considering predator risk. Estimated attack rate by Aeshna cyanea 0.095 per tadpole per hour.	CE:high SD:low DQ:high S:low C:medium E:low	Low: Laboratory experiment into prey risk assessment; extrapolation from Switzerland.	Rana temporaria	65
	Predation	MN-	Foraging success of Lestes sponsa was lower in the presence of a predator during laboratory experiments, dropping by 18.7% after the introduction of an Aeshna cyanea individual and by 25.8% for larvae without lamellae. A total of 39 out of 40 A. cyanea attacked one of the damselfly larvae (one with and one without lamellae), with 18 larvae being killed by the Aeshna.	CE:high SD:low DQ:high S:low C:medium E:low	Low: Field observations required to corroborate laboratory results as habitat selection will be important in impact of predation; larvae that were killed during this experiment were in a complex as opposed to simple microhabitat; extrapolation from Belgium.	Lestes sponsa	66
	Predation	MO-	Investigation into herbicide impact on predatory activity of Aeshna cyanea. Across experiments A. cyanea predated on a median of between 37.5% and	CE:low SD:low DQ:medium S:low C:medium E:low	Low: Laboratory experiment into impact of herbicide exposure; extrapolation from Hungary.	Rana dalmatina	67

			75% Rana dalmatina tadpoles.				
	Predation	MO-	Aeshna cyanea dragonfly larva impose a strong mortality pressure upon Lestes sponsa damselflies with daily survival probability around 10% lower in the presence of a free ranging A. cyanea compared to when A. cyanea was caged or absent. Size was also reduced for the free-ranging A. cyanea treatment, creating a significant apparent selection differential that is likely due to predator-induced reduced growth.	CE:medium SD:low DQ:high S:low C:medium E:low	Low: Field observations required to confirm laboratory experiments and determine if there are any population level impacts of A. cyanea predation; extrapolation from Belgium.	Lestes sponsa	68
	Predation	MO-	The presence of Aeshna cyanea in field enclosures reduced Lestes sponsa survival by 68% compared to treatments in which it was absent or not permitted to forage on L. sponsa damselflies. The mean head width and mass of survivors was lower in the presence of A. cyanea suggesting that larvae grew less in the presence of a free-ranging predator.	CE:low SD:low DQ:medium S:low C:medium E:low	Low: Experiment conducted in field enclosures and survival rate measured, however predation risk by A. cyanea was manipulated so it is difficult to quantify potential impact; extrapolation from Belgium.	Lestes sponsa	69
Anax imperator	Indirect impacts through interactions with other taxa	MC-	In response to caged predators, Pelophylax lessonae delayed metamorphosis more than P. esculentus, but they both metamorphosed heavier. These differences suggest the	CE:medium SD:medium DQ:high S:low C:medium E:low	Low: Laboratory experiment into the effect of A. imperator dragonfly larvae on competition betwen tadpoles of two frog species.	Pelophylax lessonae	70

			possibility of a competitive disadvantage for P. lessonae in the presence of predators, which could lead to trait-meditated indirect effects. However, the presence of predators did not modify competitive effects and had no measurable consequences on terrestrial survival.		Competitive effects and survival investigated but no measurable impacts found; extrapolation from Switzerland.		
	Predation	MN-	Anax imperator larvae killed 88.9% tadpoles when hungry and 74.1% when satiated, posing a markedly higher risk on tadpoles when hungry. Tadpole anti-refactor behaviour was less effective when predators made a greater hunting effort.	CE:high SD:low DQ:high S:low C:high E:low	Low: Laboratory experiment on predator-avoidance behaviour with hungry and satiated prey/predators; difficult to assess population level impact; extrapolation from Switzerland.	Pelophylax lessonae	71
	Predation	MN-	The foraging activity of Ischnura elegans larvae was significantly reduced in the presence of a free-swimming predator, Anax imperator. Growth was also significantly reduced, indicating that damselfly larvae adjust their behaviour in personae to predators at the expense of reduced growth and development.	CE:medium SD:low DQ:medium S:low C:medium E:low	Low: Behaviour and performance were altered under laboratory conditions, however, difficult to determine potential impact or population level effects; extrapolation from Switzerland.	Ischnura elegans	72
	Predation	MN-	Kairomones from A. imperator significantly reduced the consumption of mosquito larvae by Crocothemis erythraea, consuming a mean of 5.6 mosquito larvae in six hours when exposed	CE:medium SD:low DQ:medium S:low C:high E:low	Low: C. erythraea responded strongly to A. imperator in both laboratory and field studies in terms of feeding, however, tropical mosquito prey	Crocothemis erythraea	73

			to chemical predation cues compared to a mean of 10.4 for the control.		was used, and so similar impact on feeding rate of UK's mosquito species difficult to extrapolate from this study based in Oman.		
	Predation	MO-	Of the predators of pike fry, Anax imperator was one of the most voracious killing between 10 and 50% fry depending on fry age and killing over 43% fry for all other than the largest fry (30 days old).	CE:high SD:low DQ:high S:low C:high E:low	Low: Laboratory experiment on pike fry predation of a number of species in relation to fry size; extrapolation from France.	Esox Lucius	74
	Predation	MO-	Survival of Enallagma cyathigerum and Platycnemis pennipes was significantly reduced in the presence of Anax imperator predators compared with controls. For P. pennipes mortality was highest with A. imperator than the other predators tested, Orthetrum cancellatum and Gasterosteus aculeatus (Pisces).	CE:medium SD:low DQ:high S:low C:medium E:low	Low: Small scale field experiments in two ponds; extrapolation from Germany.	Enallagma cyathigerum; Platycnemis pennipes	75
	Predation	MO-	Anax imperator caused a statistically significant reduction in the number of Culex pipiens surviving to pupal stage (32.4%), but did not significantly effect size at or time to pupation.	CE:high SD:high DQ:high S:low C:medium E:low	Low: Artificial pool experiment measuring predation impact; extrapolation from Israel.	Culex pipiens	76
Calopteryx splendens	Hybridisation	MN-	Hybrids of C. splendens and C. virgo occur in the wild, however, estimation of prevalence indicates that hybrids are rare, constituting only ca. 0.1% of the population. This low prevalence suggests that	CE:high SD:low DQ:medium S:medium C:medium E:low	Low: Prevalence of hybrids assessed with inference that they are maladaptive, however, impacts not assessed in study; extrapolation from Finland.	Calopteryx virgo	77

			hybridisation is maladaptive.				
	Hybridisation	MN-	Contrary to predictions, field experiments revealed that territorial C. splendens males, i.e. males with the superior reproductive tactic, had the greatest propensity to hybridise with the heterospecific females (C. virgo), suggesting possible benefits. The discrepancy, however, between the number of observed heterospecific matings, and the number of observed hybrid offspring in the wild, indicates that females are either not using the heterospecific sperm or that hybrids are partially unviable.	CE:low SD:low DQ:medium S:low C:low E:low	Low: Low coherence in findings so difficult to determine impact level, negative impact inferred and not well supported by evidence; extrapolation from Finland.	Calopteryx virgo	78
	Hybridisation	MN-	Courtship attempts toward heterospecific C. splendens females increased significantly from sympatry to allopathy, suggesting allopathic C. virgo males have partially lost their ability to discriminate against heterospecific females. Therefore, when species come into secondary contact due to range expansions or range shifts, lost species discrimination ability might lead to an increase in the number of heterospecific copulations and the rate of hybridisation.	CE:low SD:low DQ:medium S:medium C:medium E:low	Low: Study design does not allow for evaluation of impact level of potential increased hybridisation between C. virgo and climate tracking C. splendens.; extrapolation from Finland and Sweden.	Calopteryx virgo	79

Libellula depressa	Predation	MC-	Predation of the mayfly Cloeon dipterum by Libellula depressa in the laboratory in a number of behavioural experiments.	CE:medium SD:low DQ:medium S:low C:medium E:low	Low: Field experiments and observations required to corroborate laboratory results; species used as model prey, other prey not included and impact not assessed; extrapolation from Italy	Cloeon dipterum	80
	Provision of trophic resources (through predation)	ML+	Experiment on behavioural traits of prey species. Three predatory fish species had three hours to prey on larvae in an aquarium with 20 larvae of one species and one predatory fish. Mortality rate of Libellula depressa was 20-30% for perch (perca fluviatils) and rudd (scardinius erythrophthalmus) and less than 10% for gudgeon (gobio gobio).	CE:medium SD:low DQ:medium S:low C:medium E:low	Low: Low mortality compared to other prey species (second lowest of five Odonates tested); experimental results; significance to predator diet not assessed; extrapolation from Germany	Gobio gobio; perca fluviatils; scardinius erythrophthalmus	81
Platycnemis pennipes	Provision of trophic resources (through parasitism)	ML+	Oviposition by p. pennipes was observed and stems collected, for these parasitism was an average of 3% and maximum of 29%. Parasitoids were identified as Anagrus obscurus.	CE:high SD:low DQ:medium S:low C:medium E:low	Low: Extrapolation from Italy; small scale study with observation of 11 individuals; parasitism measures but impact difficult to evaluate.	Anagrus obscurus	82
Sympetrum striolatum	Predation	MO-	Results reveal a high predation rate of Sympetrum striolatum naiads on the ostracod Eucypris virens. In several replicates all 20 ostracods had been eaten in the first 2 hours and considering the whole experiment	CE:high SD:low DQ:high S:low C:high E:low	Low: Potential for impact as predation rates were high in laboratory experiments, however impact on species performance and population not	Eucypris virens	83

			(360 individuals over 19 replicates) almost two thirds (n = 233) were eaten during the first 2 hours.		assessed; extrapolation from Spain.		

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