Social and ecological feasibility of a European wildcat *Felis silvestris* reintroduction

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

Tom Dando

Abstract

The restoration of species is increasingly seen as important for conservation though it is often contentious. To be delivered successfully, species reintroductions require in-depth knowledge and effective planning and assessment of their social, ecological and practical feasibility.

In this thesis, I conduct social and ecological research towards assessing the feasibility of reintroducing the European wildcat (*Felis silvestris*). Wildcats have been regionally extinct from England and Wales for over 150 years, and their reintroduction provides a compelling case study with which to explore and address key gaps in knowledge. I achieve this by taking a multi-disciplinary approach towards key issues including best-practice in social feasibility, stakeholder acceptance, site suitability, domestic cat interactions/hybridisation, and long-term population viability. Together this thesis provides an evidence base from which to inform and develop the next stages of reintroduction planning for the wildcat, as well as providing insight and direction to practitioners interested in reintroduction and conservation more broadly.

I begin by conducting the first global review of translocation literature to assess if, how and when assessments of social feasibility are conducted and use this to propose best practices in their implementation. I find that fewer than half of translocations conducted between 1922 and 2019 assessed social feasibility, and those that did tended to conduct narrow assessments focused on quantitative accounts of community attitudes. I argue that more comprehensive social feasibility assessments conducted early in planning could help address conflicts and improve outcomes, but that barriers, including insufficient expertise and prioritisation of ecological over social factors, persist. Successful projects are shown to make long-term commitments between implementing organisations, affected communities, and partners across sectors, improving resilience and outcomes. This work informs approaches to translocation planning as well as the use of the IUCN reintroduction guidelines.

I then conduct two in-depth analyses, using semi-structured interviews, of two key stakeholder groups in the reintroduction of wildcats; farmers and cat owners. For both stakeholder groups, knowledge and awareness of wildcats as a native species is shown to be low, resulting in perceived costs and benefits

being exacerbated and often not grounded in evidence. More broadly, the sampled farmers emphasise a distrust toward conservation, driven in part by a perceived anti-farmer narrative pursued by a few individuals in the media. Moreover, conservation is viewed as removed from the landscapes and communities it seeks to engage, with interviewees highlighting the value of face-to-face interactions and having an accessible presence as important to developing relationships. Among cat owners, I find a lack of consensus over who is, or should be, responsible for unowned cats. I propose the need for collaboration among a broad group of stakeholders to develop management strategies for unowned cats in the context of wildcat restoration and suggest a focus on cat welfare in communicating this. These interviews provide important insight into wildcat restoration but also key topics in conservation, namely, the perspectives of farmers on conservation practices and consequently the impact of this on conservation delivery, and the management of unowned cats. The findings of these two chapters can be used to inform approaches and topics for further engagement.

The final chapters deal with ecological aspects but are still grounded heavily in social dimensions. First, I use corridor modelling and circuit theory to analyse connectivity between woodland patches to identify suitable release areas and candidate release sites across Wales and South West England. I conduct an Analytical-Hierarchy Process to assess and rank potential release sites based on key social, ecological and practical criteria. I determine West Wales to offer the greatest potential to support a wildcat reintroduction, due to having the largest connected landscape and multiple suitable release sites. Moreover, potential threats from roads, domestic cats and conflict with rural land use are fewer when compared to South West England and North Wales. This work is instructive for practitioners interested in a wildcat reintroduction; however, I emphasise the need for additional ground-truthing. The mixed method approach used can serve as a useful methodology for any reintroduction programme to consider.

Next, I conduct single and multi-species occupancy models and temporal analysis to explore how habitat and competitive interactions influence the spatial ecology of domestic cats across three contrasting landscapes. I find that co-occurrence between cats and wild mesocarnivores is modified by habitat, but the influence of covariates differs between sites and species pairs. I observe significant spatiotemporal overlap between domestic cats and hybrid cats in Scotland, while increased densities of sheep, poultry and gamebird holdings are found to reduce co-occurrence between domestic cats, hybrids and wildcats. Results suggest changes in wild mesocarnivore occupancy and habitat conditions are likely to influence the spatial behaviours of domestic cats and consequently interactions between domestic cats and wildcats.

Finally, I conclude by discussing the contribution of this work to wider research and knowledge. This research delivers an integrated foundation to determine wildcat reintroduction feasibility through combined ecological and social assessments. It emphasises interdependencies between human and species persistence. Outcomes present a template for evidence-based decision-making enabling controversial species restoration initiatives.

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Author’s declaration for co-authored manuscripts

Chapters Two, Three, Four, Five and Six have been published, submitted or written for publication as co-authored papers with writing led by myself. I developed the research design for Chapters Two, Three, Four, Five and Six principally with Robbie McDonald and Sarah Crowley, with additional support from Steve Carter and Richard Young. Robbie McDonald, Sarah Crowley, Steve Carter and Richard Young reviewed and provided comments on all manuscripts. For Chapters Two, Four and Five, I collected and analysed all data.

In Chapter Three, Huw Denman helped to identify research participants in Wales, conduct Welsh-speaking interviews and translate transcripts from Welsh to English.

Huw Denman along with Carl Jones helped with Chapter Six by providing introductions to landowners in Wales to situate camera traps on their land. Additionally, Huw Denman provided camera data collected from Bryn Arau Duon, a woodland he manages. Roo Campbell provided camera trap data from Scotland collected as part of the Scottish Wildcat Action project to be included in the Chapter Six analysis, as well as providing review and comment on the manuscript.

Chapter One: Introduction

Ecosystem restoration in human-dominated landscapes

If current trends in biodiversity loss persist, the outcomes for human and nonhuman life could be significant (Dirzo et al., 2022). A collapse in biodiversity and the complex ecological interactions it supports alters the regulation and function of ecosystems (Cardinale et al., 2012; Estes et al., 2011; Heleno et al., 2020). The loss of functional ecosystems can reduce yields of crops, fisheries and timber (Cardinale et al., 2012; Mendenhall et al., 2014) and increase the emergence of infectious pathogens (Morand, 2020; Schmeller et al., 2020; Williams et al., 2021). Furthermore, biodiversity loss limits the ability of ecosystems to sequester carbon and regulate climate (Daba and Dejene, 2018; Díaz et al., 2009). The restoration of ecosystems and species is therefore of fundamental importance and has been recognised as such, with the UN declaring 2021–2030 the Decade on Ecosystem Restoration.

Ecological restoration was historically conceptualised as a purely ecological endeavour. However, over the past 30 years, definitions have recognised it as a socio-ecological and often political practice (Brunckhorst, 2011; Fischer et al., 2021; Higgs, 1994; Hobbs, 2004; Suding et al., 2015). In a socio-ecological context ecological restoration can be defined as “the process of assisting the recovery of a degraded, damaged or destroyed ecosystem to reflect values regarded as inherent in the ecosystem and to provide goods and services that people value” (Martin, 2017). Acceptance of a socio-ecological definition of restoration requires a concurrent shift in practice, with a focus on multi and inter-disciplinary actions to improve outcomes for biodiversity in the face of increasing human pressures (Bennett et al., 2017a; Gold et al., 2006).

Land modified by human activities has increased to 95% of global land mass (Kennedy et al., 2019). Consequently, opportunities to protect and conserve viable populations of many species in their natural habitat have declined. Moreover, many ecosystems depend on human management to persist. These transformative global social-ecological changes provide challenges for restoration (Fischer et al., 2021). Firstly, human activity is central to some of the primary threats facing ecosystems, including, human encroachment

(Dobrovolski et al., 2011; Li et al., 2022), the killing of wildlife (Sainsbury et al., 2019; Yusefi et al., 2022), human-wildlife conflict (Nyhus, 2016) and climate change (Abrahms et al., 2023), as well as habitat conversion and/or fragmentation (Li et al., 2022). Secondly, practitioners of ecological restoration must recognise the potential negative or conflicting effects it can have on multi-functional landscapes, as well as meet expectations of delivering social benefits (Fischer et al., 2021; Suding et al., 2015). Ecological restoration within human-dominated landscapes is therefore emerging as a crucial activity for delivering resilient and coexisting socio-ecological systems (DeClerck et al., 2010; Drouilly and O'Riain, 2021; Kremen and Merenlender, 2018).

Species reintroductions

Species reintroduction has often been pursued as an approach to restore species to favourable conservation status and re-establish ecosystem processes (Figure 1.1) (Polak and Saltz, 2011; Seddon, 2010; Seddon et al., 2014). Species reintroduction is defined by the IUCN/SSC (2013) as "the intentional movement and release of an organism inside its indigenous range from which it has disappeared". Reintroductions form part of a spectrum of conservation translocations, which include reinforcement, assisted colonisations and ecological replacements (Figure 1.1) (Seddon, 2010). Conservation translocations have proliferated in the 21st century (Brichieri-Colombi and Moehrenschlager, 2016; Novak et al., 2021; Seddon and Armstrong, 2016). Consequently, the evidence base surrounding best practices has also grown (Evans et al., 2023; Seddon et al., 2007; Taylor et al., 2017).

Well-planned reintroductions have been shown to provide both ecological and conservation benefits (Novak et al., 2021; Seddon et al., 2014). Research also points to an increasing 'success' rate of reintroduction programmes (Morris et al., 2021). However, reintroduced species can have actual or perceived socio-economic impacts which need to be addressed, yet which are often secondary considerations (Seddon et al., 2007). Socio-economic as well as organisational aspects of reintroductions can influence project outcomes depending on how well they are integrated into programmes (Bubac et al., 2019). Similarly, the management of projects, specifically inadequate planning, is a common factor in translocation failure (Bubac et al., 2019).

The IUCN '*Guidelines for reintroductions and other conservation translocations'* are an attempt to standardise and communicate best practices in reintroduction projects (IUCN/SSC, 2013). This guidance has been complemented with specific guidelines relating to countries and taxonomic groups (Hollingsworth et al., 2014; Linhoff et al., 2021). The IUCN guidance splits projects into three primary stages: 'Feasibility and design', 'Implementation' and 'Post-release' (IUCN/SSC, 2013). Research has predominantly focused on the latter two stages, often centred around species ecology post-release (Evans et al., 2023), however, the importance of well-founded planning and feasibility, in particular around the social side of projects, is being recognised as an area for greater attention than it currently receives in conservation practice (Ban et al., 2013).

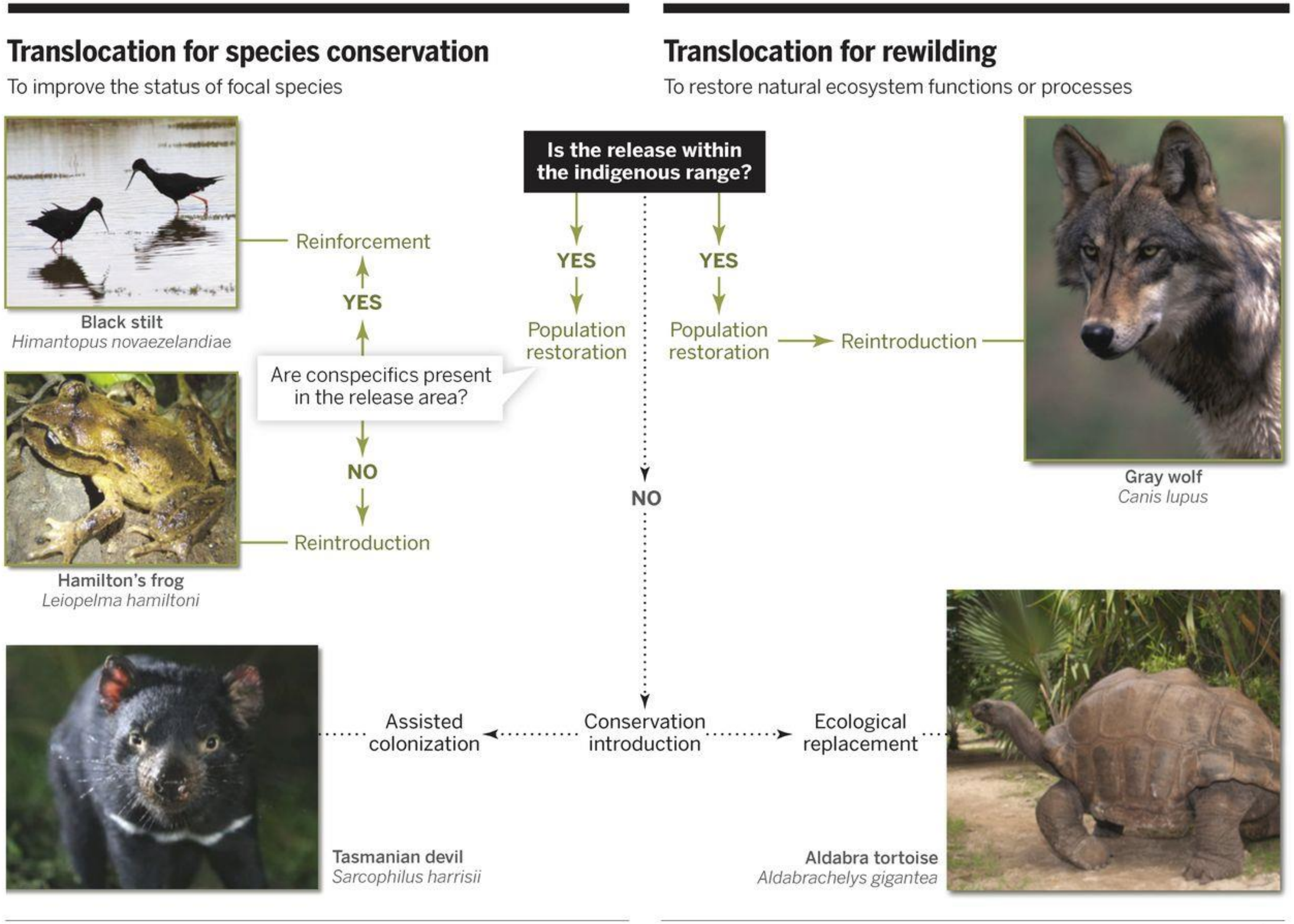

Figure 1.1 Infographic from Seddon et al (2014) of the conservation translocation spectrum based on the IUCN/SSC (2013)

Rewilding

Conservation translocations are increasingly being driven by or associated with rewilding (Figure 1.1) (Seddon et al., 2014). Rewilding has been put forward as central to ecological restoration efforts and as an optimistic vision (Donlan et al., 2006; Jepson, 2022; Svenning, 2020). Defined as the reduction of human control to return an area to a wild state (Corlett, 2016; Svenning, 2020), it is an approach to restoration which is open-ended, seeking to restore self-regulating, diverse and complex ecosystems (Svenning, 2020). Having initially centred on the concept of cores, corridors and carnivores to restore functional areas of large wilderness (Soulé and Noss, 1998), rewilding has evolved as interest in it has increased. For example, rewilding is no longer a concept solely focused on large wilderness, with the term used to encompass a variety of projects on a spectrum which occurs at a multitude of scales (Carver et al., 2021). In addition, some ‘forms’ of rewilding now focus on marrying ecological restoration with agricultural productivity (Gordon et al., 2021; Vogt, 2021). In this context, the discussion around conservation translocations in rewilding has shifted from large carnivores to including species across the trophic levels which are seen to significantly impact ecosystems (Bakker and Svenning, 2018; Mittelman et al., 2022; Zamboni et al., 2017). This evolution of what rewilding encompasses, and discussion around potential impacts are the dominant themes in the rewilding literature base, which is largely made up of essays and opinion papers. Data on the actual impacts of rewilding activities are rare but nascent in the past few years (Bakker and Svenning, 2018; Hart et al., 2023).

In common parlance, rewilding is often used synonymously with ecosystem restoration and species reintroduction (Hayward et al., 2019; Seddon et al., 2014). Rewilding can be viewed as a form of ecosystem restoration, but whereas ecosystem restoration covers a broad range of practices which usually involve human management (Anderson et al., 2019; Wortley et al., 2013), rewilding places greater emphasis on the non-human autonomy of both species and ecological processes (Anderson et al., 2019; Bakker and Svenning, 2018; Prior and Ward, 2016). Similarly, while reintroductions may be driven by the restoration of natural processes within rewilding initiatives, reintroductions also occur as a result of increasingly diverse motivations (Bakker and Svenning, 2018; Hayward et al., 2019; IUCN/SSC, 2013; Seddon et al., 2014). The extent to which these practices should be viewed as intertwined has been a frequent source of debate (Anderson et al., 2019; du Toit and Pettorelli, 2019; Hayward et al., 2019; Jørgensen, 2015; Prior and Ward, 2016). However, it can also have real-world consequences, specifically in landscapes where the idea of rewilding is controversial, such as influencing public support for species reintroductions (Bavin et al., 2020).

The popularisation of rewilding and its journey into a common language is not without controversy. The idea of restoring natural processes and reducing human control can put rewilding at odds with the custodians of managed land (Lorimer et al., 2015). Moreover, the perception of rewilding is often that it is focused on the restoration of large carnivores, creating disputes in affected communities over potential conflict (Sandom and Wynne-Jones, 2019). Finally, it has been suggested that rewilding may cause the erosion of cultural heritage from rural areas, by replacing traditional methods of land management and the communities that have depended on them for generations (Jones, 2022; Wynne-Jones et al., 2018). Consequently, many advocate a more socially focused, consensual and ethically responsible approach to rewilding, in particular within human-dominated landscapes (Deary and Warren, 2019; Drouilly and O'Riain, 2021; Lorimer et al., 2015; Thulin and Röcklinsberg, 2020). Rewilding is therefore a symbol of hope and a catalyst for innovation for some, while a source of fear for others (Sandom et al., 2019).

Species reintroductions in Britain

Britain is an interesting case study with which to explore reintroduction activities. Firstly, the extinction of many native species, largely as a result of intensive land use and killing of wildlife (Langley and Yalden, 1977) is combined with a current boom in discussion and action around candidates for reintroduction. Secondly, this is occurring in the context of a densely populated, highly agricultural island, going through significant political change toward environmental and agricultural policy as a consequence of Brexit (Helm, 2022; Sandom and Wynne-Jones, 2019).

Reintroduction and reinforcement projects in Britain have covered a wider range of taxa, including, great bustard *Otis tarda*, red kite *Milvus milvus*, ladybird spider *Eresus sandaliatus*, large blue butterfly *Phengaris arion*, Eurasian beaver

Castor fiber and pine marten *Martes martes*. There have also been unlicenced covert reintroductions including Eurasian beaver (Coz and Young, 2020; Crowley et al., 2017a; Jones et al., 2013), polecat *Mustela putorius* (Solow et al., 2006), wild boar *Sus scrofa* (Goulding, 2013), and goshawk *Accipiter gentilis* (Jones et al., 2013). Covert releases are often conducted by individuals frustrated by the licencing requirements or the speed at which such projects are accepted (Jones et al., 2013; Thomas, 2022). It can be argued that such covert reintroductions can in some instances force the agenda and create an appetite for subsequent official releases (Thomas, 2022). However, covert releases also represent challenges for conservation due to their lack of regulation, monitoring and accountability, as well as the potential for unintended consequences, such as disease transmission and human-wildlife conflict (Coz and Young, 2020; Crowley et al., 2017a; Novak et al., 2021; Ricciardi and Simberloff, 2009; Webster et al., 2006). As well as actual reintroductions, a large proportion of literature contains ecological and social research and essays around the potential for large charismatic or controversial species such, as the lynx *Lynx lynx* and wolf *Canis lupus* (Wilson., 2004; Nilsen et al., 2007; Hetherington et al., 2008; Lipscombe et al., 2018; Ovenden et al., 2019; Hawkins et al., 2020; Gwynn and Symeonakis., 2022; Bavin et al., 2023). In response to the growing interest in reintroductions in Britain, guidance for reintroductions based on the IUCN guidelines has been produced for both England and Scotland (Department for Environment, Food & Rural Affairs, 2021; Hollingsworth et al., 2014), aiming to further formalise the reintroduction process to ensure future projects meet a minimum standard of due diligence.

Human dimensions of species reintroductions in Britain

The human dimensions of reintroductions are particularly pertinent In Britain, as wildlife in rural spaces is dependent on a patchwork of commercial landscapes containing mostly agricultural land, forestry and areas for human recreation. In this context, the economics, culture, heritage and traditions of such practices and livelihoods and the communities that have lived alongside them for generations are pivotal to ecological restoration (Wynne-Jones et al., 2020a). However, this can cause tension with modern conservation practices such as reintroductions, which may provide a divergent vision of a landscape's naturalness, wildness and sense of place (Deary and Warren, 2017; Wynne-

Jones et al., 2020a). The strength of feeling toward some reintroduction initiatives means they can become politically challenging, in particular in the modern media landscape where opposing and supporting voices are amplified by social and online media (Niemiec et al., 2020a; Wynne-Jones et al., 2018). The reintroduction of species in Britain is therefore dependent on selecting desirable approaches to both engaging with communities and reintroducing a species, as well as navigating a complex cultural, political and media discourse.

Public support for wildlife restoration is thought to be growing in Britain, providing a catalyst for reintroduction initiatives (Loth and Newton, 2018). However, public support may not always be reflective of the support of, and relationships between, key stakeholders who are actually or potentially impacted by how reintroductions are delivered or those stakeholders who can influence reintroduction outcomes. For example, Bavin et al (2023) described a lack of trust between key stakeholders about the prospect of lynx reintroduction in Scotland, while Lipscombe et al (2018) found that proposals for lynx reintroduction in England, for some in the community, caused a breakdown in trust with the conservation groups seeking a reintroduction due to how the process was discussed. Similarly, in the case of the River Otter Beaver Trial, a project in response to a covert reintroduction, Crowley et al (2017a) describe the tension between opponents and supporters, with those opposing bemoaning the process under which the project was approved. Holmes (2022) meanwhile describes both strong support and opposition within the same communities when discussing reintroductions in Wales. Finally, Jones (2022) describes issues with a feeling of outside imposition, detrimental generalisations and divisive individuals in the breakdown in relationships between farming communities and conservation organisations during a rewilding initiative in Wales. Public support for reintroduction and wildlife restoration is desirable from a conservation perspective and maybe a driver for considering a reintroduction. However, making assumptions on the human dimensions of reintroductions based on a measure of general ‘support’ can be seen as unreliable and fails to account for the heterogeneity of views between and within stakeholder groups, as well as the strength that those views may be held.

Ecological dimensions of species reintroduction in Britain

The British landscape has undergone significant change since many proposed reintroduction candidates were lost (Fyfe et al., 2013). Even in the past century, there has been a major shift toward intensive agriculture as a dominant form of land use, resulting in the loss of natural and semi-natural habitats, and consequently habitat connectivity (Hooftman and Bullock, 2012). The landscape therefore looks and functions very differently since the last lynx (~1300 years ago) (Hetherington et al., 2006), Dalmatian pelican (~1400 - 400 years ago) (Crees et al., 2023) or wolf (~300 - 400 years ago) (Yalden, 1999) inhabited Britain.

The IUCN guidance places importance on understanding a species' habitat needs when assessing the biological feasibility of a reintroduction (IUCN/SSC, 2013). Assessments of habitat suitability, habitat use and landscape ecology are some of the most frequently studied areas of biological feasibility (Evans et al., 2023). Moreover, unsuitable habitat around release areas is a common cause of reintroduction failure (Bubac et al., 2019), while reactions such as hyperdispersal (Bilby and Moseby, 2023), or inadequately protected landscapes (Heurich et al., 2018) can also alter habitat use and project outcomes post-release. The latter is of particular relevance to Britain, where the protected areas, specifically national parks, have been shown to be ineffective for wildlife (Starnes et al., 2021) and not resilient to landscape-scale change (Cunningham et al., 2021). Given the significant and often unique alterations to habitats and species assemblages in Britain and an ineffective protected area network, predicting both the short and long-term response of a species is a major challenge to planning a reintroduction in Britain.

The IUCN also emphasises the importance of understanding potential interspecific interactions (IUCN/SSC, 2013). A reduction in species diversity and significant landscape degradation, alters or reduces ecosystem functioning (Cardinale et al., 2002; Fischer and Lindenmayer, 2007). This includes predator-prey interactions, as well as competitive interactions. In the context of Britain's carnivore communities, the loss of top-order predators and the killing and control of mesocarnivores between the 17th and 19th centuries (Langley and Yalden, 1977; Yalden, 1999) are expected to have had a significant effect on species interactions and therefore ecosystem function (Curveira-Santos et al., 2021, 2019; Knight et al., 2005; Ritchie and Johnson, 2009). Additionally, the introduction of non-native and/or domesticated species to the environment, such as the grey squirrel (*Sciurus carolinensis*) and domestic cat (*Felis catus*), are a further challenge to understanding reintroduction outcomes as they alter natural prey dynamics (Twining et al., 2022) and can threaten reintroduced species through competition (Glen et al., 2011), disease transmission (Bacon et al., 2023; Chantrey et al., 2014) or hybridisation (Tiesmeyer et al., 2020). This underlines the challenges of understanding the impacts of post-release interactions when reintroducing species into degraded or significantly altered landscapes and species assemblages. It also suggests that practitioners must base significant decisions on assumptions that won't be realised until a reintroduction has already been undertaken.

The European wildcat: ecology and conservation

The European wildcat *Felis silvestris* is a small felid, distributed throughout Europe (Figure 1.2) (Gerngross et al., 2021). Assessed as Least Concern on the IUCN Red List (Gerngross et al., 2021), the wildcat population is fragmented into four primary metapopulations; (1) Western-Central Europe, (2) Apennine Peninsula and Sicily, and (3) Eastern-Central, Eastern and Southeastern Europe, and (4) Iberian Peninsula (Gerngross et al., 2021). In addition, isolated populations exist in Scotland, and on Crete (Gerngross et al., 2021). The wildcat suffered widespread declines across Europe in the 19th and early 20th centuries which caused several localised extinctions (Gerngross et al., 2021; Hamilton, 1896; Langley and Yalden, 1977; Piechocki, 1990; Pierpaoli et al., 2003; Stahl and Artois, 1994). The declines were primarily due to habitat loss and killing by humans. The cessation of this pressure over the 20th century, has, in some areas, resulted in recolonisation, however, the conservation status at a national level remains unfavourable in many countries (European Commission, 2015). The most notable of these is the U.K., where the wildcat is listed as Critically Endangered by the national Red List (Mathews and Harrower, 2020) and the remnant wildcat population in Scotland has been described as 'Functionally extinct' as a consequence of extensive hybridisation with domestic cats (Breitenmoser et al., 2019).

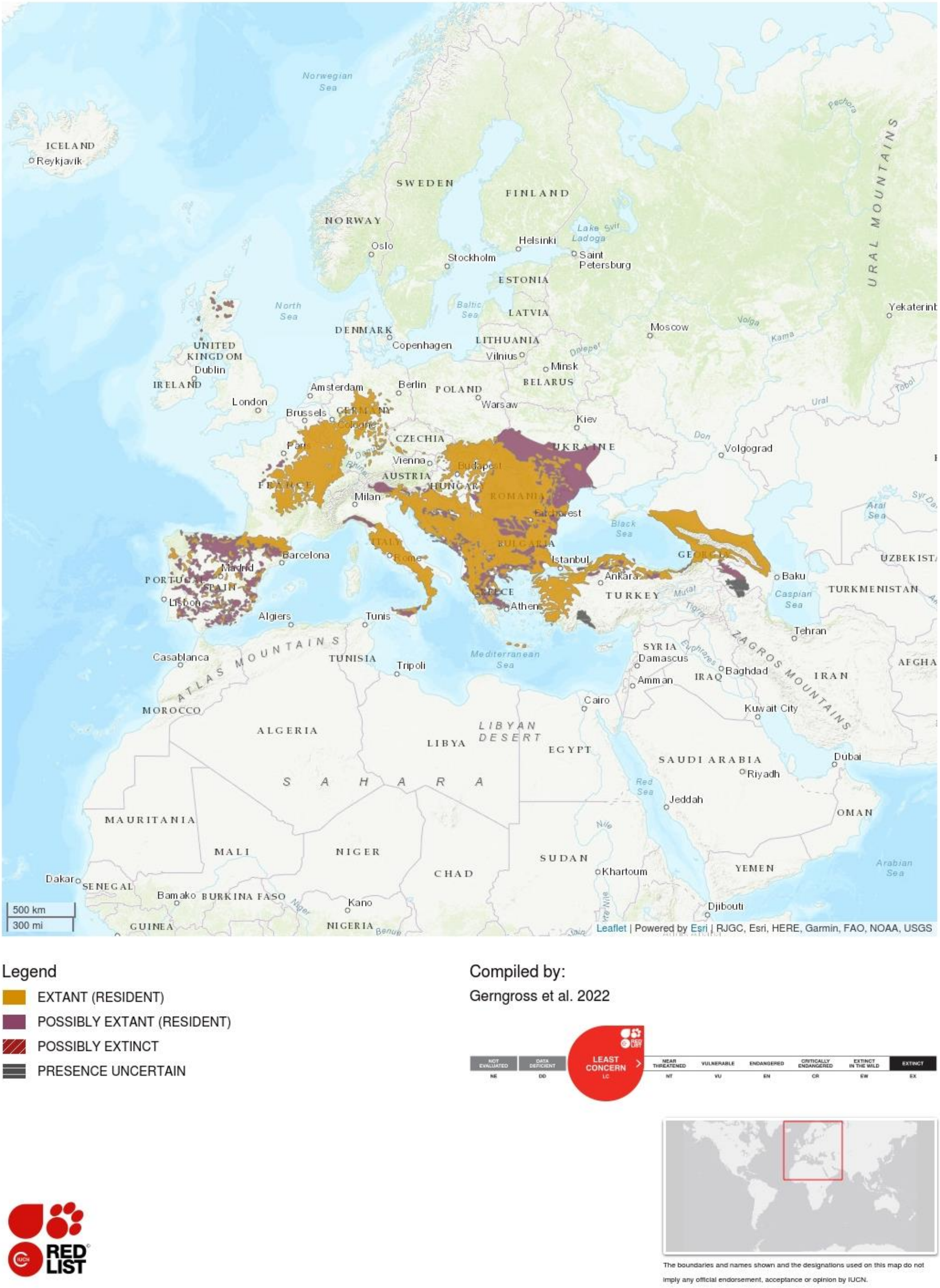

Figure 1.2 Distribution map of the European wildcat from the IUCN Red List (Gerngross et al 2022)

Contemporary threats to wildcat populations include road mortality (Bastianelli et al., 2021) and disease transmission (Bacon et al., 2023; Daniels et al., 1999), however, the literature is dominated by the threat posed by introgressive hybridisation with domestic cats (Howard-McCombe et al., 2021; Quilodrán et al., 2020; Senn et al., 2019; Tiesmeyer et al., 2020). Populations in Scotland and Hungary have reported the highest hybridisation rates, with the situation in Scotland described as a ‘hybrid swarm’ with almost all wild-living cats displaying a genetic signature of hybridisation (Pierpaoli et al., 2003; Senn et al., 2019). In other parts of Europe, hybridisation rates are lower, with estimates ranging between 3% and 21% (Tiesmeyer et al., 2020). The drivers of hybridisation are poorly understood. In south-east Europe where there are the lowest rates of hybridisation, ecological and behavioural barriers, such as competitive interaction between wild and domestic mesocarnivores, and low domestic cat densities, have been suggested as possible explanations (Gil-Sánchez et al., 2015; Oliveira et al., 2018). Fragmentation and small population sizes of wildcats have also been mentioned (Pierpaoli et al., 2003; Senn et al., 2019), in particular in the context of range expansion (Nussberger et al., 2018; Quilodrán et al., 2020).

European wildcat ecology

Wildcats are typically associated with forest habitats (Klar et al., 2008; Sarmento et al., 2006; Stahl et al., 1988), however, mosaic landscapes, including scrubland, agricultural land and riparian habitats and bodies of water have also been shown to be utilised. (Jerosch et al., 2018, 2017; Lozano, 2010; Lozano et al., 2003; Monterroso et al., 2009; Oliveira et al., 2018). Furthermore, wildcats exhibit seasonal, sex-based and individual variation in habitat selection (Beugin et al., 2016; Jerosch et al., 2018; Oliveira et al., 2018). Wildcats typically avoid human settlements (Jerosch et al., 2018, 2017; Klar et al., 2008) and areas with high agricultural intensity (Ruiz-Villar et al., 2023), in more extensive agricultural landscapes where prey density is greater, wildcats utilise linear features such as hedgerows, tree rows and watercourses, as well as small copses to hunt and travel (Jerosch et al., 2018, 2017). Human structures such as farm buildings have also been shown to be used by wildcats as resting and denning sites (Jerosch et al., 2010; Piechocki, 1990). Natural den and refuge sites include rock cairns, patches of gorse, logging piles and deadwood structures, tree roots and the crowns of trees blown over by storms (Campbell, 2015; Götz et al., 2008; Jerosch et al., 2010; Kilshaw et al., 2023; Piechocki, 1990)

Wildcat home ranges vary across their distribution, with individual male home ranges between 1.95-50.27 km^2 and females 0.69-13.85 km^2. Home range sizes and population density have been shown to reduce with increased prey availability (Anile et al., 2018; Corbett, 1979; Gil-Sánchez et al., 2020; Jerosch et al., 2017). Götz et al. (2018) found that in Germany the average annual home range size for females was 60% smaller in a rich structured agricultural landscape than in forested habitats. Male home ranges are larger than females primarily due to the roaming behaviours of males during the breeding season (Oliveira et al., 2018). Oliveira (2018) found that in Spain and Portugal, females are more likely to establish home ranges where protective cover is high, disturbance low and both prey and water sources are reliable. Moreover, females are more likely to be associated with autochthonous broadleaf forests than males, typically where rabbit abundance is high (Sarmento et al., 2006). In contrast, males are more tolerant of human disturbance and habitat fragmentation and more likely to disperse earlier and further from their natal range (Beugin et al., 2016; Oliveira et al., 2018).

Wildcats are facultative dietary specialists, specialising in rabbits where they are present and rodents where they are abundant and rabbits are absent (Lozano et al., 2006; Malo et al., 2004; Piñeiro and Barja, 2011). Moreover, at sites where rodents are not abundant, the number of insectivores, invertebrates, herpetofauna and birds in the wildcat diet increases (Lozano et al., 2006). In such circumstances, wildcats can be viewed as generalists at a local scale (Lozano et al., 2006; Malo et al., 2004).

History and future of the European wildcat in Britain

The wildcat was once widespread across Britain and is associated with many names throughout British history. These include Scottish wildcat and European wildcat, as well as historic terms, highland tiger, wood cat, British tiger and British wild cat in English, as well as cath wyllt (cat of the forest), cath y coed (wood cat), cath goed (tree cat), and cath fynydd (cat of the mountain) in Welsh and cat-fiadhaich (wild cat) in Scottish Gaelic. In old accounts the wildcat is characterised as ‘fierce’, ‘courageous’ ‘savage’ and ‘destructive’, with frequent references to its untameable nature (Clegg, 2017; Mackenzie, 1860; Pennant, 1776; Weir, 1889).

Increasing cultivation of land caused the wildcat to disappear from southern England around the 15th and 16th centuries, but they were still numerous in other parts of Britain (Hamilton, 1896). Predator control to protect game including valuable rabbit warrens between the 16th and 19th centuries, resulted in a collapse in the wildcat population (Clegg, 2017; Langley and Yalden, 1977). By the early 1800s, the wildcat was restricted to Wales, northern England and Scotland and by the mid-1800s the wildcat was thought extinct in England and Wales (Hamilton, 1896; Langley and Yalden, 1977). The last wildcat in Wales is rumoured to have been shot in Montgomeryshire in 1864, and in England in 1853 in Northumberland (Hamilton, 1896). Accounts of wildcats exist beyond these dates, however, many of these are believed to be feral cats or hybrid animals (Hamilton, 1896). By the early 20th century the wildcat was absent in Britain in all but remote areas of the Scottish highlands (Hamilton, 1896; Langley and Yalden, 1977). Hybridisation between wildcat and domestic cats has been described for centuries (Hamilton, 1896; Pennant, 1776; Weir, 1889) however, evidence suggests this was rare and had limited effect on the population (Howard-McCombe et al., 2023; Jamieson et al., 2023). Instead reproductive isolation between domestic cats and local wildcats is generally observed, likely due to behavioural and ecological differences (Howard-McCombe et al., 2023; Jamieson et al., 2023). Recent evidence suggests that this isolation has been eroded due to anthropogenic actions, with hybridisation in Scotland accelerating since the late 1950s in response to wildcat population declines, fragmentation and subsequent expansion (Howard-McCombe et al., 2021, 2023; Senn et al., 2019; Jamieson et al., 2023). Consequently, no wild-living wildcat in Scotland is free from some degree of domestic cat ancestry (Senn et al., 2019).

From the 20th to the 21st century, the wildcat in Britain has been synonymous with Scotland. Wildcats were for a time classified as a separate Scottish species, *Felis grampia* (Miller, 1907) and then a subspecies *Felis silvestris grampia* (Pocock, 1951, 1934). Wildcats in Britain have been separated from the mainland European population for 7,000-10,000 years since Britain became isolated after the post-glacial retreat of the land bridge between Britain and mainland Europe (Driscoll et al., 2007). Despite this, modern-day taxonomic assessments ascribe the Scottish population as part of the European wildcat subspecies *Felis silvestris silvestris* (Schreber, 1777) found throughout Europe (Driscoll et al., 2007; Kitchener et al., 2017), nonetheless, the ‘Scottish wildcat’ name remains commonly used.

Conservation efforts for the wildcat in the U.K. have until recently been exclusive to Scotland. While wildcat surveys occurred between 1983-1987, the first action plan wasn’t conducted until 2004. Subsequently, the Cairngorms Wildcat Project (2009-2012) and the Scottish Wildcat Action Project (2014-2019) sought to develop management strategies and reverse the declines of the wildcat. Efforts have also been made in the West Highlands of Scotland, through the ‘Wildcat haven’ an initiative with the stated aim of protecting “a naturally sustainable population of up to 1,000 pure Scottish wildcats across the West Highlands region of Scotland” (Wildcat Haven, 2023). Despite these efforts an estimated population of 115-314 individuals were determined to be present in Scotland (Kilshaw et al., 2015). Moreover, a report led by the IUCN Cat specialist group, concluded the population to be ‘Functionally extinct’ (Breitenmoser et al., 2019). The report highlighted that reintroduction or reinforcement projects would be a primary conservation tool if the wildcat was to have a future in Britain. Records and anecdotal evidence of historic reintroduction/reinforcement projects around Europe are sparse and not published and easily obtainable evidence largely absent. References exist to projects in Switzerland (Lüps, 1993), and Catalonia (Stahl and Artois, 1994), however a lack of available information on the approach taken and post-release monitoring mean lessons are hard to discern. The most well reported reintroduction took place in Bavaria where around 600 captive-bred wildcats were released between 1984 and 2009 (Büttner and Worel 1990; Mueller et al., 2020). The project initially experienced high-mortality rates, principally due to road mortality, however, was successful in that the programme was stopped when it became apparent the population had established and was expanding (BUND, 2023). In Scotland, the first releases of captive-bred wildcat into the Cairngorms National Park began in the summer of 2023 as part of the Saving

Wildcats Project, which comprises plans to release 60 wildcats over three years.

Wildcat restoration in England and Wales

In England and Wales, discussion around wildcat restoration has also been ongoing. A 2018 report by Gow and Cooper (2018) aimed to 'advise interested parties on the feasibility and practicality of reintroducing the wildcat to England' and concluded that a reintroduction would be feasible, provided associated risks are fully assessed and neutralised prior to release. The report also conducted a descriptive assessment of potentially suitable release areas, proposing Kielder Forest, the Forest of Dean or the Forest of Selwood as suitable locations. MacPherson et al (2020) used MaxEnt habitat suitability modelling to conduct a preliminary assessment of biological feasibility of wildcat reintroduction into England and Wales, seeking to 'determine whether and where further, more detailed, assessments should be focused'. The report concluded three areas should be further investigated, West Wales, North Wales and South-west England. Finally, Walsh (2020) provided an instructive commentary on how to 'implement, conduct, manage and sustain successful reintroductions for wildcats' in the UK based on discussions with practitioners working on the species in Switzerland and Germany. The report considered a wildcat reintroduction to be a 'realistic proposition' but cautioned the potential threat of hybridisation. Reintroduction remains at the planning stage with assessments of feasibility; ecological, social and practical, all being assessed, including within this thesis. In both England and Wales, partnerships of practitioners interested in wildcat restoration have been established. Presently published information on these plans is scarce.

Thesis aims and outline.

In this thesis, I use quantitative and qualitative methods to investigate the social and ecological feasibility of reintroducing the European wildcat to England and Wales. The thesis is comprised of five chapters addressing key elements of this topic, concluding with a general discussion. The thesis is split into two broad sections, with the first three chapters investigating aspects of the human dimensions of reintroductions. While the final two data chapters address the ecological aspects of a wildcat reintroduction. Despite this delineation, each chapter highlights how intimately linked the social, ecological, and practical dimensions of wildcat reintroductions are.

In **Chapter Two** of the thesis, I begin by reviewing social feasibility assessments in conservation translocations. This chapter evaluates, if and how such assessments are implemented, as well as how the assessment of feasibility influence success and failure in species translocations. Finally, I use this information to provide recommendations for best practice-related to social feasibility assessment.

Chapter Three aims to inform engagement with farmers in England and Wales by using semi-structured interviews to explore their attitudes toward wildlife conservation practices and interactions with wildlife conservation. I then build upon this by exploring current knowledge and attitudes toward wildcats and their reintroduction. This multi-layered approach allows me to understand both farmer perspectives toward wildcats and drivers of those views relating to underlying attitudes toward conservation.

In **Chapter Four,** I explore cat owner perspectives within the candidate regions in England and Wales on the impact of and responsibility for owned cats, unowned cats and wildcats. Before, examining current knowledge and perspectives towards a wildcat reintroduction. Using semi-structured interviews, I highlight the liminal position of unowned cats, and consider what this means for wildcat restoration.

For **Chapter Five,** I model habitat connectivity and core area centrality within candidate regions of England and Wales. I seek to ascertain first, if and where a connected network of habitat exists with which to support a viable wildcat population and release area, and second, I assess and conduct a decision-making analysis to compare potential release sites within the candidate landscapes.

Chapter Six uses the outputs from Chapter Five to target camera trap surveys to potential release areas in England and Wales. Using single and multi-species occupancy models, I explore both the presence of domestic cats and how their occupancy is influenced by wild mesocarnivores. I complement data from the two candidate landscapes, with data from Scotland where wildcats are already present, to also examine the role mesocarnivore assemblage plays in domestic cat occupancy and its link to hybridisation risk.

Finally, in **Chapter Seven**, I provide a general discussion of my findings and place my work in the context of both wildcat restoration in Britain and species reintroduction more broadly. I emphasise the importance of human dimensions in species restoration. Lastly, I provide recommendations for the future of wildcat restoration in England and Wales.

Chapter Two: Social feasibility assessments in conservation translocations

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Abstract

Improving the effectiveness of conservation translocations could contribute to reversing global biodiversity loss. Although evaluations of ecological factors affecting translocation outcomes are commonplace, consideration of human social factors remains rare, hindering improvements to this conservation practice. We analysed 550 translocation case studies to explore the inclusion of social factors in project feasibility assessments. Reviewed projects often failed to assess social feasibility, and assessments, where attempted, tended to be narrow in scope. Consequently, challenges such as proactively addressing conflict often remained unaddressed. Insufficient knowledge sharing and prioritisation of ecological feasibility, to the detriment of social feasibility, remain barriers to effective planning. Successful outcomes of translocations are linked to early assessment of social feasibility and to the establishment of long-term commitments between people, places, and partners.

Keywords: conservation, ecological restoration, IUCN, reintroductions, social-ecological, social feasibility

Conservation translocations are social processes

Conservation translocations are the deliberate movement of organisms, where the primary objective is a conservation benefit (IUCN/SSC, 2013). They have become an indispensable tool in attempts to reverse biodiversity loss (Godet and Devictor, 2018; Novak et al., 2021) by restoring ecosystem function (Ripple and Beschta, 2012), re-establishing and reconnecting wildlife populations (Perzanowski et al., 2020), mitigating human–wildlife conflict (Bradley et al., 2020), and as a response to climate change (Skikne et al., 2020). Despite their proliferation in conservation practice over the past 25 years, evaluations of the human social processes that often determine the outcomes of translocations remain relatively rare (Brichieri-Colombi and Moehrenschlager, 2016; Seddon and Armstrong, 2016). This is a challenge for researchers building an evidence base for effective conservation action, and highlights an opportunity for improving practice that could lead to better outcomes, both for biodiversity and affected human communities (Taylor et al., 2017). To that end, we have evaluated the conservation translocation literature with respect to the inclusion of human social factors in determining project feasibility.

Although they primarily seek ecological outcomes, translocations are inherently social processes that are influenced by organisational, political, economic, and cultural dynamics, and that exert impacts upon human communities (Crowley et al., 2017a; O'Rourke, 2014; Reading et al., 1997; Sutton, 2015) Furthermore, they are complex processes that requiring diverse knowledge to navigate stakeholder interests, convoluted funding, regulatory and logistical requirements, and challenging social-political landscapes (Berger-Tal et al., 2020; Bubac et al., 2019; Taylor et al., 2017). Social-ecological problems are often the underlying cause of the species declines that translocations seek to redress (Berger-Tal and Saltz, 2019; Dirzo et al., 2014; Fischer and Lindenmayer, 2000). The human social environment, therefore, has a major bearing on translocation outcomes, and therefore including social factors when assessing project feasibility is a crucial step in guiding decisions during planning and implementation, as well as after release, should a project be deemed feasible.

Establishing and quantifying translocation success and failure is itself a challenge (Fischer and Lindenmayer, 2000; Morris et al., 2021). Inconsistency in defining and monitoring success, and a tendency for project managers to self-evaluate, mean that quantifications of rates of success and failure have high uncertainty, and reporting is typically biased towards ostensibly successful projects (Bottrill et al., 2011; Godefroid et al., 2011; Miller et al., 2014). Furthermore, the multifaceted nature of translocations means that a project might be an ecological success but a social failure, or vice versa. In the context of assessing feasibility, the desired approach is a comprehensive assessment that aids decision making, irrespective of the outcome, although some practitioners may view the outcome as the determinant of success. This review does not seek to define success or failure; we focus instead on specific actions within translocation projects, and on whether they have been described as having positive or negative effects on project outcomes.

Biases in reporting mean that failures often go unreported and make diagnosing the causes of, and patterns in, project failure challenging (Bajomi et al., 2010; Catalano et al., 2019). This is particularly true for social aspects of translocation projects, despite these appearing to be a leading cause of project failure (Bubac et al., 2019; Catalano et al., 2019). Until 2007 the literature on conservation translocations focused almost exclusively on biology, and only 4% of studies addressed social and organisational aspects (Seddon et al., 2007). However, the basis for evaluations may have changed since that time, both in response to increasing evidence highlighting the importance of socioecological processes (Bennett and Roth, 2019), and the publication of the International Union for Conservation of Nature (IUCN) Guidelines for Reintroductions and Other Conservation Translocations (hereafter 'the Guidelines'). Published initially in 1998 (IUCN/SSC, 1998) and updated in 2013 (IUCN/SSC, 2013), the Guidelines provide an internationally recognised framework of best practices for planning and delivering conservation translocations.

Ecological specialism among conservation practitioners and a widespread (but certainly not universal) lack of expertise in, or awareness of, social sciences remain major barriers to greater attention to social aspects of conservation practice (Bennett et al., 2017a; Niemiec et al., 2021a). This is exacerbated by resource limitations, and often requires strict prioritisation of activities, typically favouring ecology, where required expertise is often already 'in-house' (Sanborn and Jung, 2021). This comes at the expense of social research, public participation, and actions that may require considerable investment to be effective, and are frequently seen to be 'nice-to-have' rather than mission-critical (Ban et al., 2013; Klein and Arts, 2022; Pooley et al., 2014).

Numerous reviews have assessed the trends, challenges, and practicalities of translocations, although these have generally drawn together ecological lessons (Batson et al., 2015; Beckmann & Soorae, 2022; Berger-Tal et al., 2020; Brichieri-Colombi & Moehrenschlager, 2016; Bubac et al., 2019; Morris et al., 2021; Novak et al., 2021; Resende et al., 2020). We shift focus here by placing such a review in the context of social lessons by utilising the Guidelines and their section on 'social feasibility'. This comprises 10 guidelines (hereafter 'social feasibility guidelines'; Table 2.1) which outline a series of organisational and socioeconomic factors that project managers should consider at the 'feasibility and design' stage of planning a conservation translocation. Although the Guidelines themselves stop short of a definition, we define social feasibility assessment as the assessment of socioeconomic factors that influence the likelihood of achieving stakeholder acceptance and realising stated conservation objectives (Crowley et al., 2017a; Mills et al., 2013; Popejoy et al., 2018).

Table 2.1 Definitions of the social feasibility guidelines for conservation translocations and the frequency of their use[a]

Guideline	Abbreviated title	Definition	Frequency of use (%)	Examples of action
1	Existing structures	Work with and/or within existing action or recovery plans, agencies, legal and policy frameworks, and infrastructure	27	Projects incorporated and learned from historic action plans and expertise when developing the project
2	Accommodate community	Plans have accommodated community, socioeconomic circumstances, attitudes and values, motivations and expectations, behaviours and behaviour changes, and anticipated costs and benefits of the translocation	35	Projects had direct contact with local communities to assess attitudes, understand local cultures, and/or encourage local participation
3	Engagement mechanisms	Mechanisms for communication, engagement, and problem solving between the public and translocation managers should be established well in advance of any release	25	Projects identified and actioned the most appropriate means of communication with the public, through either the development of a communication strategy or individual targeted actions
4	Address concerns	No organisms should be removed or released without adequate/conditional measures that address the concerns of relevant interested parties; this includes any removal as part of an exit strategy	11	Projects ensured that relevant stakeholders have a platform to voice concerns, and these could be acted upon, such as public meetings or including concerned groups in planning
5	Species connection	Where local communities may have no connection to the species or it is unknown to them, and hence oppose their release. Special effort to counter such attitudes should be made well in advance of any release	21	Projects identified target audiences and undertook awareness and outreach programmes to raise the profile of the focal species and provide information about the proposed project

Guideline	Abbreviated title	Definition	Frequency of use (%)	Examples of action
6	Economic impact	Projects should acknowledge potential positive and negative impacts on affected parties or for community opposition; where possible, sustainable economic opportunities should be established for local communities	15	Projects conducted an assessment of the costs, benefits, and opportunities, such as ecotourism, that the project could have on communities and were transparent in communicating this
7	Collaboration	Interproject, inter-regional, or international communication and collaboration are encouraged in the interests of making best use of resources and experience for attaining translocation goals and effective conservation	23	Projects created formal or informal partnerships with a diverse range of parties to maximise available expertise and resources
8	Stakeholder organisation	Where multiple bodies, such as government agencies, non-government organisations, and informal interest groups all have interests in a translocation, mechanisms for all parties to play a constructive role should be defined and the establishment of special teams that can guide, oversee, and respond swiftly and effectively as management issues arise should be encouraged	21	Projects established working groups to help to steer and structure project development and/or clearly defined the roles of implementing organisations
9	Priority alignment	Where multiple parties have their mandates, priorities, and agendas, effective facilitation should be undertaken to align priorities and resolve potential conflict areas	13	Projects undertook a process of decision making with the aim of creating an agreed plan, objectives, and direction for the project
10	Socioecological balance	Conservation actions meet the general ethical obligation to conserve species and ecosystems; however, the conservation benefits of a project should be balanced against the obligation to avoid collateral harm to other species, ecosystems, or human interests	14	Projects conducted an ethical review and/or risk assessment of the project as part of the decision-making process to identify ethical concerns, and took steps to implement solutions which minimise damage to other interests

a Frequency is based on the number of case studies that evidenced the inclusion of each guideline as a proportion of the 229 case studies where there was evidence that at least one guideline was followed during the feasibility stage of a translocation. Guideline numbers and definitions are adapted from section 5.2 (Social feasibility) of the IUCN/SSC 2013 *Guidelines for Reintroductions and Other Conservation Translocations* (IUCN/SSC, 2013)

We conducted a systematic review of conservation translocations, as defined by the IUCN, utilising the IUCN Global Reintroduction Perspectives book series (Soorae, 2018, 2021, 2016, 2013, 2011, 2010, 2008) and wider published and grey literature (details in the supplemental information online). Briefly, our review included 550 studies fitting IUCN definitions of conservation translocations, and largely excluded papers focused on species ecology. We systematically coded each study using NVivo v12. We extracted statements describing actions relevant to one or more of the 10 IUCN social feasibility guidelines (Table 2.1), and cross-coded these within categories pertaining to the social feasibility guidelines, the project stage that a relevant activity was undertaken ('feasibility', 'implementation', and 'post-release'), and whether statements were described as 'reasons for failure', 'difficulties faced', or 'reasons for success'. Our analysis identified whether, when, where and how social factors were included in reported assessments of project feasibility, and detailed barriers to their inclusion and best practices.

Inclusion and application of the IUCN social feasibility guidelines

We found that, despite evidence supporting the inclusion of social factors in translocations extending over the past 35 years (Bennett et al., 2017b; Kellert, 1985; Moon et al., 2019; Soulé, 1985; Watkins et al., 2021), this remains a minority activity. Fewer than half of the reviewed case studies reported the inclusion of social factors when assessing project feasibility and, among those that did, assessments were limited in scope; only 5% of projects included more than five of the 10 social feasibility guidelines (Box 1.1). As with other facets of the conservation translocation literature, published methodologies and evaluation of social approaches were often absent (Batson et al., 2015).

The limited scope of assessments is also apparent when looking at the frequency with which use of the social feasibility guidelines was apparent (Table 2.1). 'Accommodating community' was followed in planning more than any other guideline, typically through direct contact with local communities and assessments of attitudes towards the project. Support was frequently inferred from simple quantitative statements of majority support, indicating that some project planners view social feasibility assessments as amounting to speaking with and assessing attitudes of nearby communities (Welch-Devine and Campbell, 2010). Questionnaires were regularly used, but in many cases, it was unclear whom these targeted, or how support was measured. This raises concerns that projects are failing to consider variation in the relative influences that different individuals and groups of stakeholders will have over the feasibility and outcome of a project, as well as the importance of the relationships and power dynamics that exist between them (Felipe-Lucia et al., 2015; Ison et al., 2021).

Box 1: Trends in the inclusion of social feasibility

We analysed 550 case studies of conservation translocations from eight IUCN statutory regions, of which 419 focused on vertebrates, 85 on plants, and 46 on invertebrates. Of the case studies, 210 (38%) did not evidence any social feasibility considerations. Approximately half (259, 47%) provided evidence that at least one social feasibility guideline was followed during the feasibility stage, and 81 (15%) included at least one social feasibility guideline during and/or after implementation. For 15 (3%) case studies we cannot find evidence that they resulted in a translocation taking place.

From the 259 case studies addressing social feasibility guidelines at the feasibility stage, 42% evidenced one social feasibility guideline and 28% did so for two, whereas 5% of case studies did so for five or more, and none evidenced more than seven. There was significant variation in the frequency of inclusion of the various social feasibility guidelines ($\chi 29 = 60.8$, $P < 0.001$) (Table 2.1). 'Accommodating community' (35%) was evidenced most frequently and more than all other social feasibility guidelines, except for 'existing structures (27%) and 'engagement mechanisms' (25%). 'Address concerns' (11%) was evidenced least and was significantly negatively selected versus other social feasibility guidelines, except 'priority alignment' (13%), 'socioecological balance' (14%), and 'economic impact' (15%).

From 177 case studies, we coded 193 reasons for success into six distinct groups related to social feasibility guidelines (Table S1.1): stakeholder organisation (23%), engagement mechanisms (21%), multidisciplinary collaboration (18%), community involvement (15%), public support (12%), and conflict management (10%). From 213 case studies, we coded 232 reasons for failure into six distinct groups (Table S2): social conflict (20%), uncoordinated stakeholders (18%), opposing views (18%), under-resourced engagement (17%), political and legal barriers (15%), and communication and awareness (11%).

In total, 59% (n = 324) of case studies linked social factors to project outcomes. This varied depending on the project stage at which they were first evidenced. Of case studies that evidenced the inclusion of social factors at the feasibility stage, 37% associated these with project success, compared to 11% where social factors were absent, and 59% when their inclusion was evidenced during or after implementation. Of case studies, 27% that evidenced social factors at the feasibility stage associated them with project failure, 17% did so when none was evidenced, and 60% when the first evidence of their inclusion occurred during or after implementation.

The timing of publication of the IUCN Guidelines was associated with more frequent inclusion of social factors ($\chi 22 = 9.98$, $P = 0.006$) (Table S1.2). Case studies after the publication of the 2013 Guidelines were more likely to include social factors when assessing feasibility, and those undertaken before the 1998 Guidelines were less likely to do so. There was marked variation among the IUCN regions ($\chi 27 = 24.2$, $P <0.001$), and projects in Meso and South America (64%), and North America and The Caribbean (57%) were more likely to include social factors, whereas their inclusion in projects in Oceania (36%) was less likely than expected. All other regions fell within the expected range (Table S1.2). Significant variance was found among taxonomic groups ($\chi 22 = 20.9$, $P <0.001$); translocations involving plants (28%) and invertebrates (32%) were less likely than expected to include social factors, whereas their inclusion in projects involving vertebrates was more likely (Table S1.2). There was no significant difference among vertebrate taxa ($\chi 24 = 2.52$, $P = 0.64$).

The social feasibility guidelines followed least frequently ('address concerns' and 'aligning priorities') have much in common (Table 2.1). 'Address concerns' refers to having measures in place to address questions or challenges raised by interested parties, whereas 'aligning priorities' refers to coordinating plans and resolving conflict among primary actors (Table 2.1). Both involve acknowledging and addressing potential conflict. Identifying and addressing conflict in distinct, often unique, social scenarios is challenging given that conflicts have diverse origins (Bhatia et al., 2020; Madden, 2004). The paucity for which projects evidenced these guidelines may be indicative of the time, energy, and resources that can be required when dealing with conflict, meaning they may be set aside early in project development, especially if the required expertise is not readily available (Sutherland et al., 2010). It is also possible that early actions towards addressing conflict and aligning priorities have been treated as nugatory, low impact, or even as a hindrance to progress, creating a reluctance to plan on this basis.

Social feasibility assessment is hindered by a lack of shared best practice in how to conduct assessments most effectively (Batson et al., 2015; Sutherland et al., 2010). Hence, projects tend to coalesce around more tractable actions rather than conducting the robust, and sometimes difficult, social feasibility assessments that the Guidelines suggest are needed. This leads to more complex issues, such as identifying and addressing conflict, being side-lined. Despite the boom in translocation science, the breadth of research is often insufficient to provide evidence to support management decisions (Taylor et al., 2017). This presents a challenge to practitioners, who have access to the Guidelines but are faced with a disparate evidence base that is often disconnected from more familiar ecological research, and thus requires additional investment of time and learning (Kelly et al., 2019; MacMynowski, 2007). Research on integrating social scientific theory and methodologies into conservation planning is increasingly available (Ban et al., 2013; Bennett et al., 2017b; Niemiec et al., 2021a; Raymond and Knight, 2013) but the challenge is not only to ensure there is a framework within which to include social feasibility but also to provide an evidence-based justification for the methods of making the assessment. Without this, the application of the IUCN social feasibility guidelines is likely to be guided by individual or organisational experience, anecdote, or not at all (Cook et al., 2010; Sutherland et al., 2004).

Timing of social feasibility assessments

Ultimately, the use of social feasibility assessments will be influenced by evidence of their impact on project outcomes. Our analysis indicates that conducting feasibility assessments during project planning is linked to a reduction in the frequency of reporting social challenges during the implementation and post-release stages of translocations (Box 1). In projects where social factors were only considered after the feasibility stage, greater social challenges were reported. In these instances, actions were often implemented as a reaction to the emergence of social problems, creating unexpected resource burdens, as well as impacting on public tolerance and awareness of both the species and the project (Jeong et al., 2010; Sharp et al., 2010; Widenfalk et al., 2018). Resource limitation is universal in translocation projects, meaning that reactively allocating adequate resources to solve post-release problems is invariably challenging (Bubac et al., 2019). The timing of social feasibility assessments is therefore important (Ban et al., 2013; Sterling et al., 2017); assessments made too late could lead to projects proceeding in circumstances that an earlier assessment might have identified as severely compromising feasibility. Projects may be driven or compelled to continue because of insufficient resources to reverse releases, but ultimately the viability of the project may be affected (Watkins et al., 2021). Planning and decision frameworks are increasingly available (Schwartz et al., 2018), and participatory approaches, such as structured decision making (Gregory et al., 2012) and adaptive management frameworks, are regularly used (Brown et al., 2022). This enables project managers to break down complex problems, such as assessing whether a translocation should proceed, into smaller decisions through the analysis of different scenarios, while also facilitating the direct involvement of stakeholders (Ewen et al., 2014; Panfylova et al., 2019). Although clearly beneficial, such approaches still require the right timing and social expertise within the process to identify potential threats and design effective tests of steps in mitigation.

Social feasibility outcomes

The reviewed literature suggests that internal factors (where implementing organisations interact with each other) were as important as external factors (implementing organisations interact with external stakeholders) for project outcomes. 'Stakeholder organisation' (agreements and planning within project partnerships) was the most frequently cited social reason for success and was characterised by long-term commitments and support at local, national, and international levels. This coincided with increased resource capacity and resilience. Including a diversity of partners across sectors, disciplines, and scales also provided greater breadth of expertise to inform robust planning and implementation.

Engagement mechanisms, such as public outreach and education programmes, were the second most frequently reported reason for success. However, in most cases, the detail and evaluation of these activities and their effectiveness remained unreported. The value of such actions can be seen in the case of the recovery program for the Antiguan racer snake (*Alsophis antiguae*; Box 2). Conversely, when these activities were not prioritised and were under-resourced, they were highlighted as a prominent reason for failure, and in most cases inaction was linked to resource availability (Adams et al., 2014; Tooze and Baker, 2008). Building support and participation through engagement activities involves developing an understanding of key stakeholder viewpoints and knowledge levels (Ison et al., 2021), and subsequently the creation of appropriate framings of the project (Consorte-McCrea et al., 2022; Niemiec et al., 2020b). This is particularly true of poorly understood species and/or those that are likely to be contentious (Hiroyasu et al., 2019). In these instances, early articulation of any costs or benefits is imperative to reduce the risk that uninformed opinion or misinformation might influence attitudes (Hiroyasu et al., 2019). Even then, it is unrealistic to expect rapid consensus, as demonstrated by projects such as grey wolf (Canis lupus) reintroduction into Yellowstone National Park and Idaho, USA, where it took 20 years of debate to achieve broad support for the project (Bangs and Smith, 2008). Failing to allocate appropriate resources to support public engagement and participation in planning can be detrimental to both project reputation and outcomes (Tosi et al., 2015). Cases of social conflict commonly involved highly mobile, large species whose release generated negative public opinion and where the species were often perceived as detrimental to human interests. These outcomes are commonly a consequence of failures to take adequate measures to identify and address concerns in the planning stages, as in the case of the tammar wallaby (*Macropus eugenii*; Box 3) where conflict between communities and conservationists occurred after a largely ecological planning process failed to address social concerns.

Together, the characteristics of successful and failed translocations tell us that projects making early commitments to people, places, and partners for the long-term and taking a multidisciplinary approach represent good value and are more effective at improving knowledge, organisational relationships, and resource resilience. Committing to affected people, as much as to wildlife, is likely to lead to more successful practice.

Box 2 Case study: The Antiguan Racer Conservation Project (ARCP)

Background

The ARCP is a multi-partner project formed in 1995 in response to the immediate threats facing the species (Daltry et al., 2001). At the inception of the project, ~50 Antiguan racers (*Alsophis antiguae*) remained on Great Bird Island, Antigua (Daltry et al., 2003). Research found that this island could only sustain ~100 individuals in the long term (Daltry et al., 2003). A 10-year reintroduction action plan was therefore drawn up to reintroduce the racer to other islands in the region (Daltry et al., 2017).

Social assessment

In planning, the project partners committed to investing in building local capacity and resources with the aim of the project being run by local organisations deemed crucial for its long-term sustainability (Daltry et al., 2017). They also identified that few Antiguans or tourists knew about the Antiguan racer, and most of those who did, expressed a negative attitude, with many presuming them to be dangerous, causing them to be deliberately killed (Daltry et al., 2017). Similarly, trampling and the prevalence of campfires were identified as key issues to address (Daltry et al., 2001). Raising awareness and education were determined to be key actions if any reintroduction was to be feasible, and private landowners, tour operators, and other regular island users were identified as key stakeholders (Daltry et al., 2001).

Actions

The project team utilised a range of media and tapped into local knowledge within the project team to inform people about the racer. Activities included making television documentaries, hosting field trips, posters, presentations, and newspaper articles, as well as workshops with local guides (Daltry et al., 2001). Further to this, the Antiguan Racer Schools Campaign targeted children throughout Antigua to enable them to visit the remaining population on Great Bird Island and learn about conservation; this was accompanied by teacher training (Daltry et al., 2001; Daltry et al., 2017). In 1999 this education and engagement work formed part of the endorsement of a formalised plan between the IUCN/SSC Reintroduction Specialist Group and the Antiguan Racer Conservation Project (Daltry et al., 2017).

Outcomes

The partners of the project credited these actions as one of the primary reasons that the species has avoided immediate extinction (Daltry et al., 2003). Knowledge and opinions towards the racers showed significant improvement, evidenced by 26% of visitors to Great Bird Island first hearing of the racers through the education programme, as well as the racer becoming a symbol on the Antigua and Barbuda EC$50 telephone card and being prioritised in the National Biodiversity Strategy of these countries (Daltry et al., 2001). Finally, many Antiguan schools were reported to have established wildlife conservation in their syllabus as a result of the schools campaign and wider engagement activities (Daltry et al., 2003).

Box 3 Case study: the tammar wallaby reintroduction

Background

Plans to reintroduce the tammar wallaby (Macropus eugenii) to South Australia (SA) were submitted by the Department for Environment and Heritage in 2004 (2004). The mainland tammar subspecies had become extinct in the region in the 1930s owing to habitat clearance and fox predation (Sharp et al., 2010). It was listed for reintroduction as part of the Federal 1996 Action Plan for Australian Marsupials and Monotremes (Maxwell et al., 1996). The Innes National Park, an area surrounded by intensively cultivated agriculture and its rural community, was selected as the most suitable site for release (Department for Environment and Heritage, 2004).

Social assessment

The project put a great deal of planning into captive breeding, site selection, and post-release monitoring, greatly outweighing any social feasibility work, which made up two pages of the 66 page reintroduction proposal (Department for Environment and Heritage, 2004). According to Peace (2009), the project assumed that because the biodiversity credentials were so compelling, any concerns of local people would be easily dealt with. The project failed to identify and address potential conflict areas, although the spread of the tammar wallaby beyond the boundaries of Innes National Park was fundamental to the success of the project, and farming communities held a negative view of the implementing body before the project (Peace, 2009).

Actions

Although a public relations campaign was planned, details of the project were leaked to local farmers before this work had been conducted (Peace, 2009). In response, the community was dismayed that an animal that had been declared an agricultural pest on nearby Kangaroo Island and in New Zealand should be reintroduced; news also spread to the local media who expressed their displeasure at the plans (Peace, 2009).

Outcomes

An 18 month conflict unfolded between practitioners and residents, consuming a considerable proportion of project resources (Sharp et al., 2010). Local people were concerned that populations of red fox (Vulpes vulpes), a non-native invasive species, would not control the spread of the wallabies as scientific modelling had suggested, leading to threats to human livelihoods (Peace, 2009; Sharp et al., 2010). Retrospective consultation failed to rectify the divide, and trust between the two parties had been eroded. In the face of opposition, the relocation of the wallabies into the National Park went ahead (Sharp et al., 2010). Despite the conflict and continuing opposition from farmers post-release, a stable population has been established (Peace, 2009).

Arguably a robust social feasibility assessment would have identified many of the issues that occurred, thus allowing project leaders to plan and prioritise resources to address them. Alternatively, it may have determined that the project was socially unfeasible at that time, despite its ecological and technical strengths. The team of practitioners leading the project have stated that the primary lesson learned is that community engagement needs to occur well in advance of any reintroduction (Sharp et al., 2010).

Barriers to inclusion of social factors

Understanding what practical and institutional barriers prevent greater incorporation of social feasibility is an important step toward increasing accessibility. Our review indicates that the key barriers are (i) insufficient resources and lack of prioritisation for social scientific research and insights, (ii) lack of in-house expertise or inclusion of social scientists during planning, and (iii) differences in terminology, methodology, and literature bases that limit access to, and appropriate deployment of, robust social research methodologies.

Appropriately resourcing social scientific research would enable a robust, evidence-based rationale for wider social feasibility assessments, and would also provide insights that aid the development and targeting of engagement activities, such as education and outreach programmes. Such programmes are effective at increasing local knowledge and positive attitudes (Leisher et al., 2012); prioritising resources here will therefore benefit overall project outcomes. Resource prioritisation toward social research should also be reflected more widely in conservation science training to aid the development of practitioners equipped with the skills needed to meet present-day conservation challenges (Bennett et al., 2017a; Gardner, 2021).

The inclusion of social scientists early in the planning process is an important step in bringing social considerations to the forefront of translocation planning. Planning in wildlife management is typically led by ecologists, meaning that the identification of social issues and access to appropriate methodologies is often limited (Robinson et al., 2019). Removing such barriers enables projects to integrate social scientific thinking, which often comes from different perspectives on, and approaches to, both research priorities and the roles of people in ecosystems (Sanborn and Jung, 2021; Williams et al., 2020). Challenging an embedded way of working that is configured towards the natural sciences would be made more achievable by placing a greater emphasis on developing in-house social scientific expertise which can shift organisational cultures as well as support specific projects.

Finally, breaking down barriers in terminology and literature bases would improve communication and flow between the social and natural sciences in translocation projects (Fox et al., 2006). Such barriers limit access to and sharing of best practices and mean that learning across disciplines requires extra dedication of time. Journals are increasingly taking an interdisciplinary approach which will enable greater knowledge sharing between disciplines (Pooley et al., 2014; Williams et al., 2020). Furthermore, we would encourage the IUCN Global Reintroduction Perspectives book series to place greater emphasis on the social side of its case study reporting.

An overview of best practice

By bringing together lessons from our reviewed case studies and wider conservation literature, we can start to identify what best practice in social feasibility looks like. Although covering methodologies for every social feasibility guideline and project scenario is beyond the scope of this review, we can identify broad themes and priorities.

Establish partnerships with shared goals

We have identified early and long-term commitments to stakeholder individuals and organisations as a precursor to successful outcomes. In many cases we found that this was facilitated by formal agreements between implementing organisations, such as a memorandum of understanding or the development of an agreed long-term plan (Bridge, 2016; Freifeld et al., 2016). This process and the surrounding discussions help to coalesce partners towards shared aims, while the creation of an agreed leadership structure can aid the delineation of responsibilities and overall accountability (Sutton, 2015). These agreements work in parallel with discussions to clearly define objectives, needs for resources and skills, and costs. Taking a multidisciplinary approach has also been linked to successful outcomes through the utilisation of diverse expertise in the creation of a project plan, as well as by facilitating access to resources and local communities (Cisternas et al., 2021; Freifeld et al., 2016). Together these actions help to inform the organisational feasibility of undertaking a translocation project and identify areas where the project requires additional actions or knowledge to become a feasible endeavour.

Conduct a dedicated social feasibility assessment

At present, the inclusion of social factors in feasibility assessments appears to be sporadic and largely fails to follow social feasibility guidelines. If social feasibility is to become a standard part of the conservation translocation process, we believe projects should conduct a dedicated social feasibility assessment (a specific document that addresses each guideline in turn) at the earliest opportunity, preferably alongside ecological feasibility assessments, and certainly in advance of translocations taking place. This provides ample opportunity to explore and analyse the social landscape and enables such early assessments to shape translocation planning, rather than being dictated by prior decisions and commitments. When integrated into a structured decision-making process, this enables projects to identify relevant challenges and action appropriate solutions. Furthermore, it may be that a project is ultimately deemed socially unfeasible and therefore should not proceed. Early assessments may therefore save resources and reduce the risk of damaging relationships between conservationists and stakeholders. Every social feasibility guideline does not need to apply to every project, but by creating a standardised process, projects can explicitly justify their inclusion or exclusion, as well as providing a clear route to the evaluation of decision making post-release. Conducting assessments in this way would reduce reactive decision making and better enable project managers to prioritise actions and resources from an early stage.

Invest in community research and engagement

The social feasibility guidelines highlight the importance of ensuring that the views and circumstances of interested and affected communities are incorporated into feasibility assessments, and surveying public attitudes appears to be the most common way in which social feasibility is assessed. Although quantifying public support may provide a useful overview, projects should consider what information is most relevant to determining feasibility when designing assessment methods. Commonly used, and often relatively untargeted, approaches such as questionnaires and online surveys are rarely meaningful in isolation, and when designed without prior knowledge may ask the wrong questions of the wrong people. They may also fail to pick up the underlying reasons why particular views are held, which in some cases may not be related to the species or project at all, nor do they account for the changeable nature of public opinion, or power dynamics within and among stakeholder groups (Ison et al., 2021; Niemiec et al., 2021b). Projects could benefit from mixed-methods approaches, for example by conducting interviews with targeted individuals who may have specific knowledge or represent key stakeholders, combined with or, perhaps better, followed by questionnaires aimed at larger populations (such as residents) to understand the range and prevalence of different views (Hanson et al., 2020; Newing, 2010; White et al., 2005).

Projects should seek to bring local people and organisations closer to the project, thus providing opportunities for community participation and wider public buy-in (Brooks et al., 2013; Christie et al., 2017; Poe et al., 2014), which in turn would facilitate longitudinal monitoring of local attitudes (Niemiec et al., 2021b). A more integrated and transparent approach decreases the feeling of outside imposition and builds trust, enabling projects to develop a deeper understanding of affected communities, cultures, and traditions, and to explore drivers of support or opposition (Gregory et al., 2012). This could include both formal methodologies such as interviews, focus groups, and public meetings, as well as action-orientated approaches such as providing a platform for community participation in the design and delivery of projects. Community participation brings clear benefits to translocations (Andrews et al., 2010; Brooks et al., 2013; Consorte-McCrea et al., 2022) that are not often realised (Klein and Arts, 2022). The feasibility stage is the best time to explore the range and prevalence of different views and building relationships, and the methods of assessment should reflect this. Conducting thorough engagement as part of the feasibility process should not be viewed as attempting to make a project more or less feasible but as a means to inform better decision making, which in turn provides better long-term conservation outcomes for wildlife and people.

Address divisions and identify consensus

Translocations are multi-stakeholder endeavours. Therefore, it is crucial to encourage open dialogue and an inclusive process that acknowledges the range of viewpoints and places all stakeholders as part of the process rather than outside of it. One method to align the priorities of these wider groups is through the use of workshops or working groups (Low, 2018; Young et al., 2020). These can include a multitude of stakeholder groups and encourage wider participation in the planning process, thus helping to legitimise decision making. Inclusive approaches to decision making have their challenges (Jami and Walsh, 2014; López-Bao et al., 2017); however, ensuring broad representation of views, through open and transparent dialogue, early in planning, will highlight where consensus exists, and where division and conflict may arise, as well as providing wider context for any underlying issues (Christie et al., 2017; Coz and Young, 2020; Manfredo et al., 2021; O’Rourke, 2014; Young et al., 2020). These can be discussed and acceptable mitigation and an adaptive management framework developed through collaborative means, ahead of time (Low, 2018; Redpath et al., 2013; Young et al., 2020).

Concluding remarks

Although awareness of the 2013 IUCN Guidelines may have helped to increase uptake of social feasibility assessments in conservation translocations, these practices remain narrow in scope and largely opaque in their reporting. This indicates that translocation projects still do not generally apply a comparable focus on planning for social and ecological factors. This is due to a lack of information regarding best practices, insufficient social expertise, and resource prioritisation towards ecological and technical feasibility. A reactive approach to assessing social issues, in particular around conflict, leaves projects vulnerable to unexpected outcomes. Integrating bespoke social feasibility assessments into planning could alleviate some of these issues and provide a clear route to evaluating outcomes and prioritising actions. Furthermore, increasing institutional capacity for social scientific research and advice within conservation organisations, and in turn addressing the lack of time and funding given to social aspects of translocation projects, should be considered to increase the resilience of dynamic projects that are working in complex social-ecological circumstances. In addition, the value of making long-term commitments to translocation project partners, places, and people cannot be overstated in achieving positive outcomes for biodiversity conservation.

Chapter Three: Farmers' perspectives on wildlife conservation practices and implications for a European wildcat (*Felis silvestris*) reintroduction

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Abstract

Farmers are important stakeholders in many conservation projects. However, their relationships with conservation practices and institutions can be challenging, in part due to differing visions and priorities for the same spaces. Reintroduction of carnivores into farmed landscapes can be especially contentious, because of actual or perceived risks to livestock and livelihoods. Effective engagement between conservationists and farmers is essential for positive reintroduction outcomes. The European wildcat (Felis silvestris) is Critically Endangered in Scotland and is extinct in England and Wales, where reintroduction has been proposed. Using semi-structured interviews, we investigated farmers' perspectives on conservation practice, focusing on wildlife reintroductions and the prospect of wildcat restoration. Farmers often perceived wildlife conservation practices as removed from the needs of rural landscapes. Discourses initiated by prominent individuals and amplified in the media were perceived as 'anti-farmer' and have fostered feelings of distrust of conservation practices and associated organisations. While we highlight farmers' senses of detachment and imposition, most farmers expressed willingness to engage with reintroduction projects if they were engaged in the 'right' way. Face-to-face interactions, and investment in a long-term local presence were seen as essential in engendering positive relations between farmers and trusted individuals. Cultural salience of wildcats was low among farmers in these regions, where wildcats were long extinct. Uncertainties and confusion about wildcat ecology meant that many farmers, irrespective of their support for reintroduction, overstated both negative impacts and potential benefits. The conflation of reintroductions and 'rewilding' appeared detrimental to support for reintroductions. Transparency and clarity in communicating the scope of a project and farmer involvement were important. Individual and community level engagement as well as local involvement in planning reintroductions are central to fostering positive relationships between farmers and conservation organisations. Where the cultural salience of a species is low, such approaches can reduce the risk of misinterpretation of a species impacts and project objectives. Our wider exploration of current problems and potential solutions (as perceived by farmers) between farming and wildlife conservation mean our results apply to a host of conservation initiatives where there is a need to facilitate better interactions between these groups.

Keywords: European wildcat, Felis silvestris, Reintroduction, Social feasibility, Farming, Agricultural landscapes, Stakeholder Engagement, Rewilding

Introduction

Farmers are a prominent stakeholder group in many nature conservation initiatives, primarily due to the prevalence of farming as a form of land use in many parts of the world (Henle et al., 2008). Farmers affect, and are affected by, the trajectory of many conservation initiatives, but conservation practices cause tension when they are perceived to conflict with farming livelihoods (Balfour et al., 2021; Dando et al., 2022; Henle et al., 2008). In countries such as the United Kingdom, where 69% of land is in agricultural use (Department for Environment, Food & Rural Affairs, 2022), mobile species are likely to interact with farmland in some way. While some species are tolerated and encouraged by farmers, others are considered undesirable and often subject to management, including lethal control (Cerri et al., 2017; Drouilly et al., 2021; Horgan and Kudavidanage, 2020; König et al., 2020). Understanding the experiences of farmers concerning conservation practices and how associated interactions can be improved is fundamental to achieving positive outcomes for many conservation initiatives (Erisman et al., 2016). Using semi-structured interviews, this study seeks to ascertain how farmers in England and Wales perceive species reintroduction initiatives, with a specific focus on the European wildcat *Felis silvestris*.

Farming and wildlife conservation practices

Relationships between farmers and conservation organisations are often complex and can be contradictory. Both sets of actors may compete over separate visions for the same space and with ostensibly divergent priorities (Mikołajczak et al., 2022). Despite this perceived opposition concerning issues around the conservation of biodiversity (Henle et al., 2008), they can also be allies on issues both believe to be detrimental to the rural landscape (Sherval et al., 2018). Furthermore, the groups are not mutually exclusive: in many instances, farmers self-identify as conservationists, or as stewards of rural landscapes and their traditions (Logsdon et al., 2015; McGuire et al., 2013; Raymond et al., 2016; Sherval et al., 2018). Within farming communities, there is also significant variation in attitudes toward wildlife conservation, between those who are engaged, opposed, and apathetic (Barnes et al., 2022; Upadhaya et al., 2021). Farmer motivations for involvement in conservation projects can be both intrinsic, e.g. a love for their land and a desire to pass it on in a good condition for future generations, or extrinsic, e.g. economic benefits (Greiner and Gregg, 2011; Ryan et al., 2003). Responses to a conversation intervention also vary with farmer understanding of its predicted environmental effects (Arbuckle, 2013) and the perceived impact on their crops or livestock (Bavin et al., 2020) This variation is important to consider in tailoring engagement activities across farmed landscapes.

How individuals conceive of wildlife conservation also varies, and can place farmers and conservation practitioners at odds with each other (McEachern, 1992). Farmers can perceive conservation as a collection of institutionalised practices (Coz and Young, 2020; MacDonald, 1998; O'Rourke, 2014), diminishing the role of the individual conservation practitioner in debates around conservation and farming. A perception of conservation as institutionalised may also limit the value of bipartisan relationships at an individual level. This can have cultural connotations around who has the right to decide the future of privately-owned land, rural landscapes in general, and the methods and systems by which conservation is 'done'. Cultural tensions can, at their worst, be linked to accusations of conservation being imposed upon key actors, and of 'fortress conservation' approaches, including the exclusion or replacement of people and culture in favour of biodiversity outcomes (Garland, 2008; Holmes, 2014; Rai et al., 2021). Additionally, cultural division has been linked to differences in demographic, educational and economic factors (Manfredo et al., 2020), with perceived divides between rural and urban communities (van Eeden et al., 2019; Witt et al., 2009), lived experience and academic knowledge (Aswani et al., 2018; Garland, 2008; Rust et al., 2022; Skaalsveen et al., 2020) and economic groups (Holmes, 2014, 2011; Wynne-Jones et al., 2018).

Productive stakeholder engagement in wildlife conservation has long been discussed, however, evaluation of when and why different approaches are effective remains scarce (Chase et al., 2004; Reed et al., 2018; Sterling et al., 2017). The rise of social media and online news media adds an extra dimension to these discussions due to the extensive reach and velocity it provides for conservation messaging when compared to traditional in-person engagement (Bergman et al., 2022). Such methods, however, risk misrepresentation of messages occurring and exacerbation of conflict if online messaging occurs ahead of direct engagement with affected stakeholders (Hart et al., 2020;

Pennycook et al., 2018). Stakeholders who feel their concerns have been ignored are less likely to develop a trusting relationship. This can inhibit collaborative working and initiate or escalate disputes towards conflict. Identifying appropriate engagement methods and timing is therefore an important step in facilitating positive communication between farming and wildlife conservation.

Stakeholder engagement in species reintroductions

Species reintroductions are an important tool in conservation practice and are used in attempts to reverse local extinctions and biodiversity decline (Godet and Devictor, 2018; Novak et al., 2021). Socio-political challenges to reintroductions are a common cause of project failure (Bubac et al., 2019; Dando et al., 2022). In many cases, conservation organisations fail to address adequately the concerns of affected stakeholders and incorporate them into conservation strategies (Dando et al., 2022). This increases the chance of conflicts with the reintroduced population and also erodes trust between conservation institutions and other stakeholder groups (Madden, 2004). In Britain, public debates around species reintroductions and rewilding often appear polarised in the media (Marino et al., 2023; Wynne-Jones et al., 2020b). Reconciling these debates through engagement between key stakeholders is fundamental to ensuring the needs of rural industries and wildlife conservation are met, and that modern conservation practices are fit for purpose in agricultural and other human-dominated landscapes.

The need for 'renewed coexistence’ created by the reintroduction of predators to farmed landscapes can be a significant source of social conflict, as farming communities are required to adapt to lost and often forgotten species (Auster et al., 2021a; Banasiak et al., 2021; Bavin et al., 2020; O’Rourke, 2014). Species lost before living memory often suffer from ‘societal extinction’, with negative consequences for perceived cultural connections and both scientific and experiential understanding of species ecology (Jarić et al., 2022). Reintroductions can help reinvigorate a species' cultural presence, but in the absence of direct knowledge and experience, deep-rooted environmental and political values, which can be challenging to change, are just as likely to drive individual opinion (Hiroyasu et al., 2019). This is especially true for predators, which are often a source of tension between conservationists and farming communities (Hawkins et al., 2020; O'Rourke, 2014). Species reintroductions into farmed landscapes, therefore, require the restoration of a species to both the physical and the socio-cultural environments.

In contrast to mainland Europe, recolonisation by terrestrial species that have been driven to national extinction is unlikely or impossible on islands such as Great Britain, without human intervention. Several high-profile reintroductions and reinforcements have occurred in Britain, such as beaver (*Castor fiber*) (Crowley et al., 2017a; Gaywood, 2018; Jones and Campbell-Palmer, 2014), white-tailed eagle (*Haliaeetus albicilla*) (Dennis et al., 2019), white stork (*Ciconia ciconia*) (Dempsey, 2021) and pine marten (*Martes martes*) (MacPherson and Wright, 2021). Investigations into the feasibility of reintroducing other high-profile species are ongoing (Bavin et al., 2023; Hawkins et al., 2020; MacPherson et al., 2020). As a consequence, there is a growing literature on social dimensions of species reintroductions in the UK (Alif et al., 2023; Auster et al., 2021a, 2021b, 2021c; Bavin et al., 2023, 2020; Coz and Young, 2020; Crowley et al., 2017a; Marino et al., 2023), which can be utilised to improve outcomes for the reintroduction of other lost species, particularly those that may be deemed controversial. This work identifies the significance of stakeholder engagement for project success, and highlights the need to explore and integrate concerns and areas of potential conflict stakeholders may have into reintroduction projects.

European wildcat reintroduction

Wildcats became extinct in both England and Wales in the mid-1800s, largely due to historical predator control (Langley and Yalden, 1977). The only remnant population in Britain is in Scotland, which has been declared 'functionally extinct' due to hybridisation with domestic cats (Breitenmoser et al., 2019; Senn et al., 2019). The conservation status of this species and its isolated distribution in Britain mean that it is a national conservation priority, with reintroduction into suitable areas of its previous native range in England and Wales now being considered, alongside reinforcement in Scotland (Breitenmoser et al., 2019; Campbell et al., 2023a; Gow and Cooper, 2018; MacPherson et al., 2020).

While often thought of as a woodland species, wildcats have been shown to exploit landscapes with a mosaic of habitat types, utilising open, often agricultural land and linear features such as hedgerows to hunt and move through landscapes (Jerosch et al., 2017; Lozano et al., 2003; Monterroso et al., 2009). As a result, they are likely to use farmland to a significant extent in any landscape to which they are reintroduced. Wildcats are facultative dietary specialists, and mainly eat small mammals and rabbits (*Oryctolagus cuniculus*), depending on their availability (Apostolico et al., 2016; Lozano et al., 2006; Malo et al., 2004; Ruiz-Villar et al., 2022). Wildcats do not typically represent a threat to livestock, however predation of livestock, particularly poultry, while rare, is a potential source of conflict (Lozano et al., 2006). Given the length of time that wildcats have been absent from England and Wales, knowledge of their appearance, behaviour and ecology among stakeholders is likely to be low. Consequently, we expected farmers' perceptions of the wildcat to be influenced by their perspectives more generally on conservation practices and reintroductions. Understanding these broader value orientations, alongside responses to the wildcat, can highlight additional factors that may shape support for reintroductions, as well as key concerns that can be used to inform engagement strategies. As a mesocarnivore, which would be expected to utilise farmed landscapes, the potential reintroduction of the wildcat represents an informative case study with which to explore farmers' views on (a) conservation and its practices, (b) species reintroductions, and (c) wildcat reintroductions specifically.

Methods

This exploratory study used semi-structured interviews to explore farmers' perspectives on conservation organisations, species reintroductions, and on wildcats. This qualitative approach allowed us to explore themes at a greater level of detail when compared to large-scale surveys and has been used effectively in similar studies which sought an in-depth understanding of stakeholder perspectives on wildlife management (Coz and Young, 2020; Crowley et al., 2018; Klein and Arts, 2022; Mikołajczak et al., 2022; Swan et al., 2020). Interview questions focused on three main themes: (a) What is the current relationship between farming and wildlife conservation practices and why? (b) What do positive and negative interactions between farming and conservation look like? And (c) How can this knowledge be used in the implementation of a species reintroduction? We use the wildcat as a specific and timely case study through which to explore these broader questions.

Farmers were recruited and interviews conducted between November 2020 and June 2021. Recruitment used both stratified random sampling and snowball sampling (Newing, 2010). We did not seek a truly representative sample, instead favouring qualitative semi-structured interviews and thematic analysis methods which capture detailed perspectives, while still giving sufficient variance across the sample to address our research questions. However, by combining sampling strategies we ensured our sample included a cross-section of farmers. We targeted our recruitment to farmers in south-west England and south-west Wales, which have been identified as regions to investigate the potential reintroduction of wildcats (MacPherson et al., 2020). We focused recruitment on poultry and sheep farmers specifically, as these were most likely to be affected, or to perceive themselves to be affected, by a wildcat reintroduction. Our primary method of recruitment was through direct contact. This involved compiling a list of farms within our target areas using information derived from Google Maps. In the first instance, we emailed farms, making a phone call where contact numbers were publicly available, and an email address was not found. All emails contained an information sheet detailing the study and what participation would entail. Farms were informed that participation was voluntary and that they had the right to withdraw at any time.

Our snowball sampling involved asking participating farmers if they could recommend farmers in their network for us to contact. In addition to participating farmers, we asked key local informants to help identify farmers within our study regions. All Welsh interviewees were given the option to have their interview conducted in either Welsh or English and all interviewees signed an informed consent form before the interview took place. 22 farmers participated in the study. Numeric coded identifiers are used here to protect participants' identities.TD conducted all English-speaking interviews and HD all Welsh-speaking interviews. Interviews followed a semi-structured schedule. This approach gave interviewees the freedom to discuss and expand upon issues they deemed relevant or connected (Young et al., 2018). Interviews were conducted in 2021 and 2022 using both video and telephone calls due to Covid-

19 restrictions. The study received ethical approval from the University of Exeter's CLES Penryn Ethics Committee (eCORN003107)

All interviews were audio-recorded and transcribed verbatim. Interviews conducted in Welsh were translated into English by HD for analysis. Interviews were then analysed using NVivo (v12). Transcripts were thematically coded in three stages by a single coder (TD). First, transcripts were individually coded by grouping interview responses by questions and combining answers into four coded groups: (a) relationship with wildlife conservation practices; (b) interactions with wildlife conservation (c); perspectives on species reintroductions; (d) perspectives on wildcats. Second, codes within the resulting groups were separated into broad themes based on the subject matter of the content. Finally, the content of each theme was sub-coded to combine related perspectives within each theme. This process allowed us to group responses and identify the diversity of perspectives prevalent in our interviews and their relevance to our primary research questions.

Results

Wildlife conservation practices

Erosion of trust

Farmers' perspectives on current conservation practices were characterised by a lack of trust (Figure 3.1). This lack of trust was felt by farmers to be mutual, with farmers frequently describing a breakdown in trust with conservation groups, but also expressing a feeling that conservation groups do not trust farmers. Concerning the former, some interviewees described broken promises, dishonesty, and anti-farmer rhetoric, typically through news media.

"The media narrative around these schemes [species reintroduction and rewilding] has damaged relations between farming communities and conservation. It's eroded trust and a lot of people just think why we would want to engage with people like that" [08]

For some, it wasn't necessarily that conservation NGOs themselves were delivering anti-farmer messages. Instead, they were perceived to stay silent or support what were considered extreme views espoused by a small number of individuals and a misrepresentation of the role of UK farming in biodiversity declines and climate change. Several interviewees said that farmers were at the mercy of government priorities and had simply done what they have been asked to do.

“What I see within the conservation sector is because of the popularity of this sort of charismatic messaging, and rewilding narrative, the conservation sector just stayed quiet. And as a result, that's sort of classic complicity. And that's added to the erosion of trust… There's a wider community distrust of some of these organisations now, and part of that is because they're seen as being tacitly dishonest by not calling out some of the techniques that some of those at the ego-driven end are using” [17]

Only one interviewee characterised this relationship as improving.

“I have seen a positive change; I’ve seen organisations come and stand next to the farmers which maybe back in 2000 didn’t happen” [10]

Role of the media

The role of the media in this discourse was prevalent throughout and associated with polarising the relationship and giving a voice to what were considered to be extreme views (Figure 3.1). This was not only related to conservation but also to farming communities, where interviewees felt the good work done for wildlife by farmers was not represented in the media.

“I think a big part of it is to treat farmers with more respect. A lot of people are tired of being painted as the villain. There is a lot of good stuff happening in the farming community for nature, but it gets ignored” [08]

Lack of contact

When asked about direct interactions the majority of interviewees could not recall being involved with or speaking directly to conservation groups (Figure 3.1). In the absence of direct contact, the strength of feeling has built through online and media communication, or through second-hand information shared in farming communities. Instead of listing conservation bodies, however, farmers listed a handful of prominent individuals as being catalysts for the erosion of trust and communication between farming and conservation, as well as what was described as the ‘rewilding movement’.

"*Division sells these days, so it's in the media interest to stoke conflict in the countryside and I think that's what he* [prominent environmental commentator] *thrives on and makes his money from. He puts himself and rewilding forward as some silver bullet and farmers as the barrier to eutopia. It has been harmful and a factor in the widening divide between farming and conservation"* [21]

We also saw evidence of a cultural divide within this discourse. Farmers often portrayed conservationists and government agencies as being removed from the rural landscapes which they seek to change. This implies the divide is one not just of what a landscape should look like but also the feeling that in many cases conservation is imposed from the 'outside'.

"You only see conservationists when they want something. It's easy to sit in your heated office and make plans to rewild, but these are people's lives and I think if they spent more time in these communities and came and spoke to us and listened to the problems we have, the issues that their schemes might cause, then there is an opportunity to have more positive outcomes" [08]

Value orientation

Farmers placed greater value on lived experience, local knowledge, and 'common sense', and in some cases derided scientific and academic arguments as not being grounded in the "*reality of the countryside*" (Figure 3.1). In the instances where this was not the case, the participant tended to have come to farming via other avenues or had gone through university education.

"*Lots of people who have been in this work for generations will keep going regardless and they need to, all that knowledge and history, it would be criminal if that was allowed to disappear in favour of a countryside where decisions are made from urban academics… They place too much value on books and not enough on lived experience"* [06]

Rewilding

The term 'rewilding', and approaches associated with this were not part of our primary question set, however, dominated many of the discussions and in almost all cases - regardless of the interviewee's views on wider conservation or environmental issues - the focus on rewilding was viewed as damaging to relationships between farming and conservation (Figure 3.1). Only one interviewee expressed unequivocal support for rewilding and its advocates though they acknowledged that their livelihood was not dependent on their farm, while others suggested that aspects of rewilding could be incorporated into farming practices and that both sides were guilty of a failure to compromise. Two interviewees had not heard the term ‘Rewilding’ before.

“Not all, but many [conservationists] *don’t understand the reality on the ground and get caught up with environments commentators and the rewilding lot, which is a tricky place to be for farmers and working together, because there is a view that rewilding is anti-farmer, and I feel like all the hype and pushing of that narrative distracts from the serious conservation issues we are facing that farmers and conservation groups should be coming together on”* [02]

As a conservation approach, rewilding was also viewed as the antithesis of how a landscape should be managed and what it should look like. Additionally, it was commented by a small number of interviewees that a managed landscape is what people want to see.

“The reason that people enjoy this area; they think it’s an accident that it’s like it is; that’s people’s perception. The reality is that it’s managed; a lot of work has gone into making it like it is so that they can enjoy it as it is. Rewilding would mean a lot of negative consequences” [04 translated from Welsh]

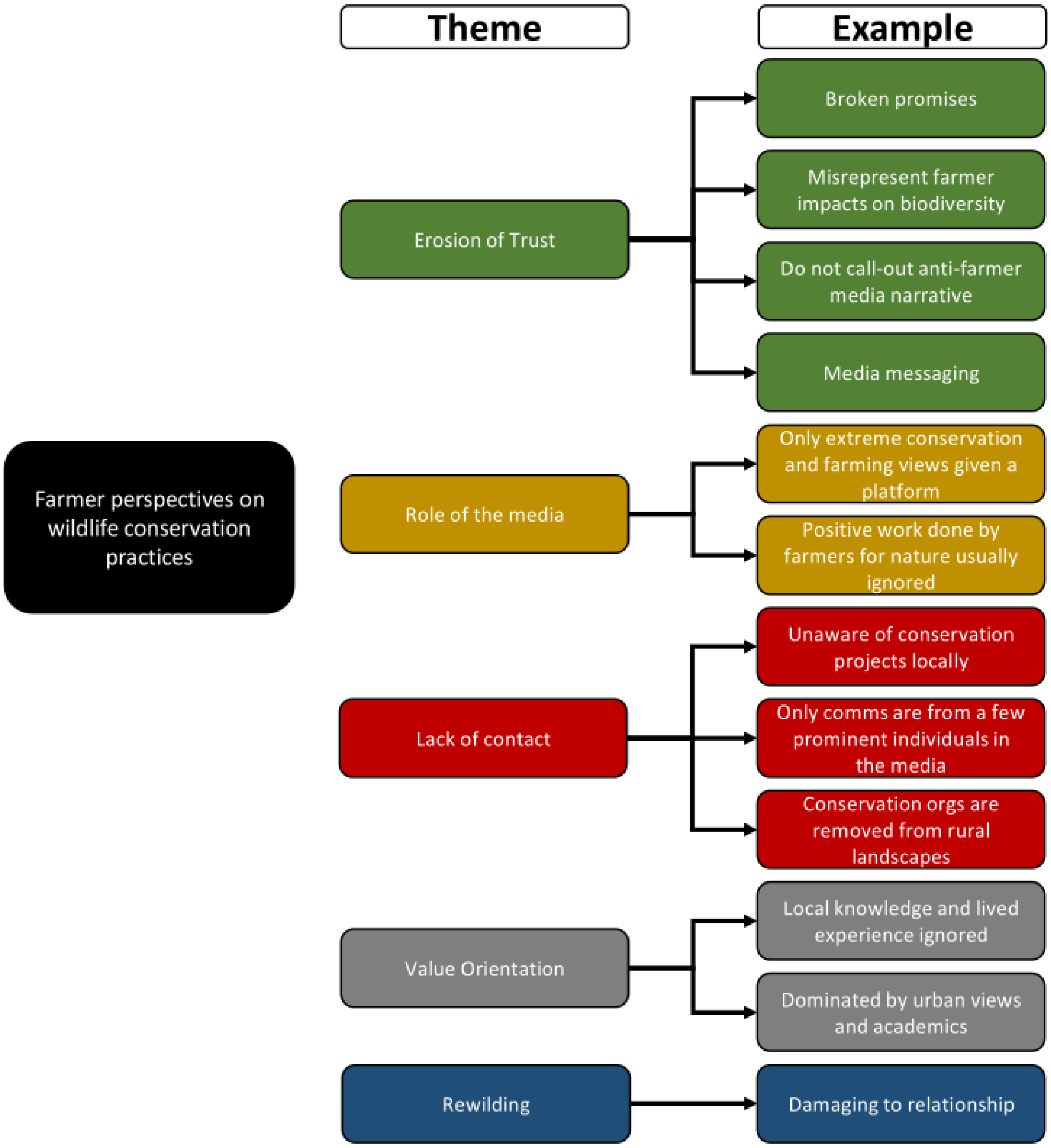

Figure 3.1 Themes of farmer perspectives on wildlife conservation practices and common examples given by participants related to each themes.

Species reintroductions

In-person engagement

The most prominent point raised was that any interaction should occur face-to-face (Figure 3.2). This was perceived to assist relationship-building and transparency, provide a known point of contact for local communities to ask questions, and avoid miscommunication. Many felt it much easier to find common ground, or at least develop a respectful relationship, through face-to-face dialogue, rather than when interactions occur online or through the media. The reinforcement of pine martens in Wales was cited as a good example of this, with several interviewees living in the locality of that project highlighting the positive influence that the project team's approaches had on community responses.

"the community felt like we had questions answered and they [Vincent Wildlife Trust] *seemed to answer them honestly in that they didn't always know, but they could show us the research and were willing to engage"* [02]

Place and community based

Farmers indicated they would respond better and be more willing to engage if messaging was coming from the local community, local groups, or respected individuals therein (Figure 3.2). Disdain for the outside imposition of reintroduction and conservation projects was apparent throughout most interviews. One frequent example provided by Welsh interviewees was a landscape-scale rewilding initiative in Wales, described by one interviewee as

"*A typical modern conservation project, tied up with all this rewilding nonsense. All faceless and run by people who don't own or live on the land, but feel they know what is best for our communities"* [08].

Furthermore, several farmers suggested that they would be more inclined to listen to farmers from other locations (where the proposed species was already present), as trusted sources of information than they would conservationists.

Integrated into local cultures, customs and language

Specifically, in Wales, the use and inclusion of the Welsh language, as a way of ensuring that local cultures and customs form a central part of any conservation project was highlighted (Figure 3.2). Moreover, the role that language plays in being able to understand local structures and engage fully with local history was emphasised. The need for projects to go beyond translating key documents towards genuine integration and involvement of Welsh language and Welsh speakers was evident if a project is to feel and belong to Wales.

"They [Conservation organisations] *arrive and assume that there's no civic environment, that there are no structures in place for discussion, or that there's no sort of cohesive community view, because they simply can't hear it. They don't understand it when it's expressed as it is expressed in a different language. And the civic structures around the language are different to the civic structures that you get around the English language".* [17]

Emphasise co-benefits

Another area highlighted by farmers was the need for conservation organisations to promote farming and do more to emphasise the co-benefits of projects for both farming and conservation goals (Figure 3.2). Interviewees expressed that they felt the good work that many farmers do is often ignored, with conservationists choosing to use farmers as a means to highlight biodiversity challenges, rather than working together to find solutions.

*"[*Conservationists should be*] finding common ground, supporting the work of farmers rather than constantly criticising it, and talking face to face rather than through unions, or the media. Spend time here and get to know the place and its people and find a solution that benefits all parties"* [06]

Accountability

The next area of discussion was around accountability (Figure 3.2). Several farmers were concerned that once a reintroduction occurs, nobody would take responsibility for any negative impacts of the species which had been reintroduced, or that there would be insufficient monitoring, which could lead to farmers being blamed for negative effects. Much of this was connected back to the need for a transparent plan, a point of contact, and a clear exit strategy, with accountability clearly stated.

"*There must be an exit policy and management plan agreed upon right at the beginning of the reintroduction program. I'd feel the same with anything that was native that we want to bring back that there must be clear lines of* *accountability and a very robust exit strategy or plan. If you've got an offending animal; beaver, marten, wildcat or whatever, there needs to be a straightforward process to remove it if it's causing a problem, whether it's live catching or lethal method*" [22]

Continuity of personnel

Continuity was also described by some as an important part of developing relationships and interactions (Figure 3.2). The high turnover of staff and the short-term nature of contracts within many conservation organisations were highlighted as a hindrance to long-term collaboration and trust. Reintroductions are often sensitive projects and having continuity of individual staff members was seen as a benefit, allowing local communities to get to know and trust practitioners and vice versa.

*"The [*conservation organisation*] people appeared, did a survey and said they'd be back, and we never heard from them again. What they lack is continuity of staff. People come in for a year or two and then disappear to do other things, or their funding runs out"* [03]

All outcomes are an option

Ensuring that engagement occurs with an open outcome was mentioned by several interviewees (Figure 3.2). Even concerning examples which were frequently expressed as being positive, such as the pine marten reinforcement to Wales, it was pointed out that although the engagement was carried out well, there was still the feeling of a pre-determined outcome, which lessened the desire for communities to communicate and engage.

"Initially, when we're talking about the pine marten reintroduction, it was presented as a fait accompli*. And then it was discussed. So there wasn't really an option not to introduce pine martens on the table"* [17]

Communicate objectives

Finally, It was apparent, that for some farmers, reintroductions have become conflated with rewilding and apex predators (Figure 3.2). Therefore, given the concerns about 'rewilding' highlighted in the previous section, the first response of some interviewees was to push back against types of projects that were perceived to negatively impact farming or be intended to replace farming. For others, however, while negative perceptions of rewilding remained, reintroductions were seen as seeking a specific, tangible outcome, rather than landscape-scale change. This led to more balanced discussions around reintroductions, with many interviewees broadly positive about them. However, despite a lack of opposition to the idea of reintroductions, farmers were largely critical of the approach conservation organisations have taken to implementing such projects.

"Generally positive about reintroductions. But I do think that these programmes can cause conflict… I think there is an ethical problem for me to be planning a reintroduction project where you would expect the species to present in the farmed landscape and not involve any farming interest in the planning and operation" [11]

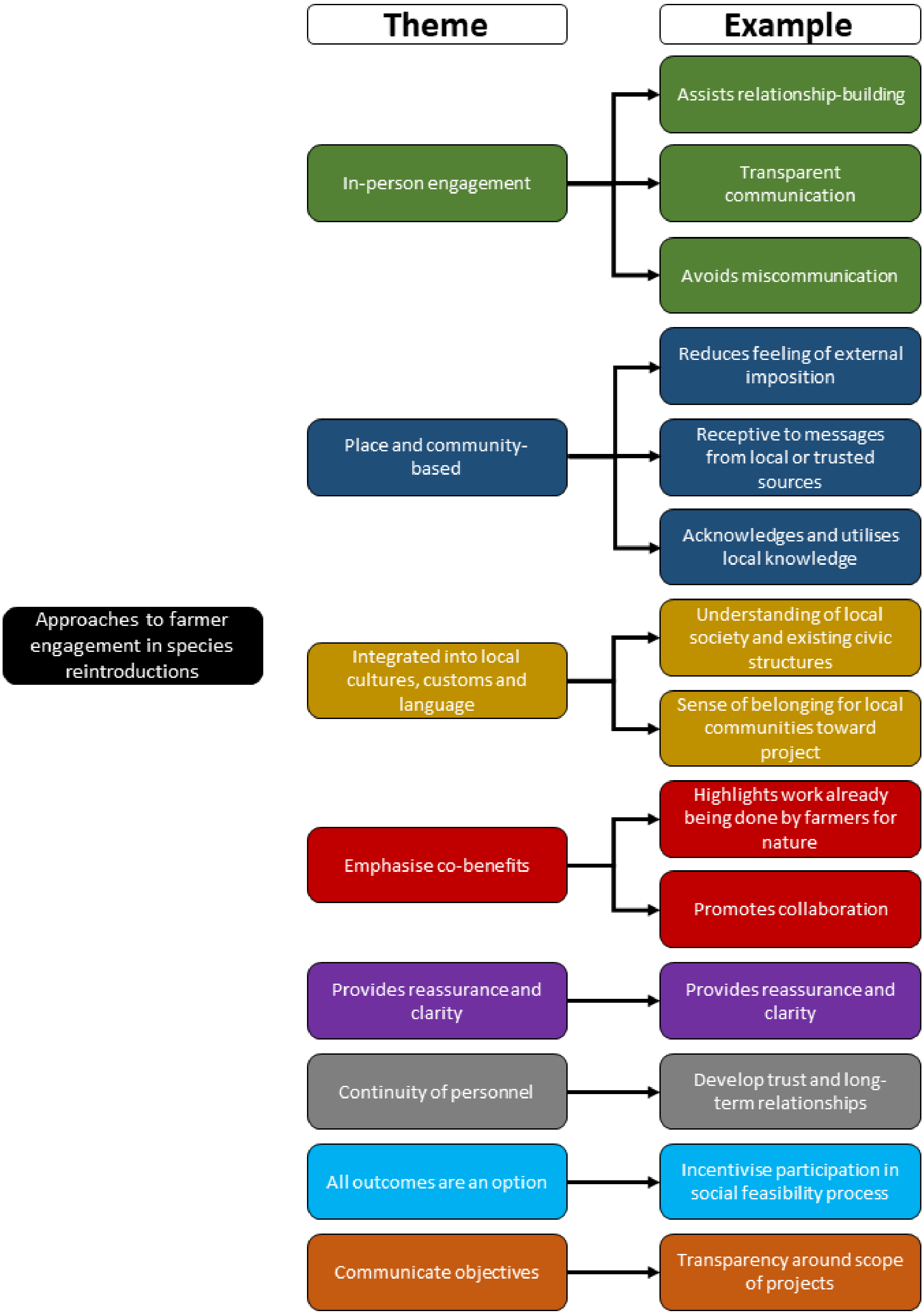

Figure 3.2 Themes of positive engagement between farmers and conservation organisations in species reintroduction projects and common examples given by participants related to each theme

Wildcat reintroduction

Interviewees expressed a spectrum of views on the potential reintroduction of wildcats to England and Wales (Figure 3.3).

I would support a reintroduction if

Many interviewees had concluded that the wildcat posed no threat to livestock and had the potential to help with the control of species, such as grey squirrels (*Sciurus carolinensis*), and mice (*Mus musculus*), as well as provide competition for red foxes (*Vulpes vulpes*). Consequently, they saw no reason why a reintroduction should not occur. A small group of farmers also felt there was a moral obligation to at least consider a reintroduction if a species is native and their extinction was caused by humans. Others said they would support a reintroduction, but that wildcats would need to be controlled should their population go on to have a negative impact.

*"Some people might assume they [*wildcats*] are bigger or might be a threat to sheep but if they had the information and could see them, then I doubt many would oppose. If they take a few greys [*grey squirrels*] or help bring down fox numbers and bring a bit of balance, then that can only be a good thing"* [02]

I neither support nor oppose a reintroduction

A second group, while echoing some of the sentiments of those in favour (such as not being concerned about interactions between wildcats and livestock), expressed hesitancy and reluctance to either support or oppose. A lack of information was the main reason for this, with most suggesting they would be receptive to additional information before deciding. Additionally, how this information is disseminated was also raised, with a personal approach, such as face-to-face conversations preferred to indirect communications.

"it would be good to have all the information and like we've said, someone has a face-to-face chat and says look this is why this is happening and this is what the benefits would be and an honest appraisal of any impact it might have" [06]

Among this group, there was also a feeling of being unconvinced by the necessity and likely success of such a reintroduction. This manifested itself as concerns over hybridisation and the number of domestic cats in the environment, as well as the ecological niche that wildcats inhabit, overlapping with owned and unowned (feral) domestic cats that are already present. 'Priorities' was an important term for this group, and while they did not demonstrate any major opposition, it was felt that the resources, time and energy of a reintroduction project should be concentrated elsewhere on projects that have more obvious and timely benefits to farmers and the landscape.

"*I don't have any major opposition to the idea of wildcats. But the money that goes into these projects and I imagine everything you'd need to do to remove all the moggies* [domestic cats] *out there isn't cheap. I just feel could be better used elsewhere. All those resources going towards an initiative with co-benefits to wildlife and farming could transform a landscape. I guess I see wildcats as nice to have but we have bigger issues at the moment"* [20]

I oppose a reintroduction

The final group all opposed a wildcat reintroduction but fell into two subgroups. The first were opposed not due to wildcats, but rather due to their ideological views on landscape, wildness, and conservation more broadly which are inherently opposed to reintroductions. Rewilding and reintroduction were commonly conflated by these farmers.

"It would be a waste of time [wildcat reintroduction]. This is not a wild landscape, people work, live and enjoy it as it is. Our land has been managed in a certain way for centuries… Conservation groups don't know how the countryside works or what it needs. [21]

The second subgroup was opposed to wildcat reintroduction more specifically, based on their understanding of wildcat characteristics, ecology, and impacts – such as estimates of their size and the size of prey they might take. Many of their concerns ran counter to the current scientific understanding of wildcats, including, for example, concerns that wildcats might pose a threat to people.

"I wouldn't want to see the wildcat back as maybe we'd have trouble with them with people and children… There's some reason that they've disappeared. Either there was a disease they had or something. The dinosaurs have gone. There you are. Amen!" [07 translated from Welsh]

Wildcat reintroduction was also questioned by one interviewee who queried if a wildcat was 'natural' to the area due to its long absence. Suggesting that if that is the approach, then why not reintroduce other 'unnatural' species.

"If you bring something [wildcats] to this area that's unnatural, you just as well bring a lion here then. It's not natural in the area and hasn't been here for centuries" [04 translated from Welsh]

Across all three groups, clarifying questions were frequent (Figure 3.3). Common themes included both the positive and negative effects of wildcat predation, the size of wildcats and responsibility for them post-release. The frequency of such questions indicates that while current knowledge is limited, interviewees were curious about the species and reintroduction process.

Figure 3.3 Summary of farmer responses to the reintroduction of the European wildcat to their area. Divided into four groups: supportive statements, neither support nor oppose, opposition statements and clarifying question

Discussion

Our research highlights growing discursive divisions between farmers and the institutions and practices of wildlife conservation. Farmers cited an anti-farmer media narrative, with which conservation organisations were considered complicit, as the primary driver of this division. In addition, conservation institutions were viewed to be largely absent from direct engagement with the rural landscapes they seek to change, hindering the development of relationships and allowing distrust to develop. Nonetheless, most Interviewees expressed a willingness to engage in conservation activities. Farmers highlighted the benefits of localised presence and personalisation of conservation practice, through face-to-face interactions and engagement with, and ideally, involvement of, affected farmers and local farming communities. Together, this suggests that the divide between these two groups is not necessarily about the people doing conservation, but rather the current institutionalised nature of conservation structures and practices. While we found a spectrum of views on the specifics of a wildcat reintroduction, from support to opposition, a key driver of opposition is founded in wider ideological views of landscape and countryside management, rather than concerns about wildcats specifically. Additionally, our results indicate the methods used by conservationists to engage with farmers could have a significant impact on the level of support for a reintroduction. Finally, dissociating any proposal from an unpopular ‘rewilding narrative’ and some of its exponents, is seen as important in building relationships with farmers.

Farmer-Conservation interactions

The extent of farming in the UK means that effective conservation is largely dependent on collaborative approaches and environmentally sympathetic approaches to land use. Balancing wildlife restoration, food security and the economics of the countryside are important to sustainable farming and preserving biodiversity. The perceived distrust and poor relationship between farmers and conservation institutions found here and elsewhere (Bavin et al., 2023; O’Rourke, 2014) and the perception that conservation practices are often inaccessible to farmers are therefore likely to be counterproductive to achieving conservation goals and wider environmental protection.

Many farmers in this study were unable to identify a conservation project with which they had been involved, or that had happened locally. Given the UK's extensive and well-established network of conservation organisations and wildlife recorders (Lorimer, 2015), this is surprising and suggests that beyond the delivery of projects, conservation institutions often don't maintain a presence within the landscapes they hope to conserve or restore. Furthermore, it implies that modern conservation practices often occur at a distance from people who work in the countryside. Our interviews indicate that this adds to the feeling of external imposition and difficulty in building relationships. Moreover, it emphasises a growing cultural divide, with conservation institutions' voices perceived to be less valid in determining the direction of rural landscapes as a consequence. This absence means that farmers' perspectives on conservation projects and conservationists themselves can be affected by a small number of individuals prominent in the media (Thomas, 2022). When describing positive interactions with conservationists, farmers invariably described in-person dialogues, with open and transparent conversations. In many instances such interactions were informal. This highlights the value that trusted individuals have in delivering conservation. In contrast, divisive messengers and the misrepresentation of messages in the news media and social media can damage community perceptions of species and projects (Bergman et al., 2022; Hart et al., 2020; Wynne-Jones et al., 2018). The relationship between farming and conservation appears to be defined as much by failures in communication as by conservation ambitions or actions. Often the issues that create tension may not be linked to conservationists or projects, but to how they are conducted. Even those who broadly support the ideas of species reintroduction or rewilding were often critical of how such projects have been approached.

Many farmers appear to be suspicious of external expertise, instead seeking knowledge and learning from networks of peers (Rust et al., 2022; Skaalsveen et al., 2020). Where trusted sources of information come from, draws on a wider sense of culture and belonging and a fear of outside imposition, with science and often conservation organisations viewed as removed from a sense of place and context, and disconnected from lived experience (Bavin et al., 2023; Jones, 2022; Satterfield, 2007; Walsh, 1997). Conservationists were often characterised as urban, elite, and removed from local knowledge. This characterisation is not a recent phenomenon (e.g. McEachern, 1992) and feeds into a narrative in which some farmers correlate their lived experience with increased knowledge about nature and conservation and that because agriculture created modern British landscapes, only farmers know what is best for them. Tensions are therefore increased when farmers' experiences are overlooked in favour of scientific evidence in decision-making (Harrison et al., 1998). Within more formalised structures, having a local presence within project planning and disseminating messages, was important to building trust. In the context of reintroductions, farmers are willing to listen to messages from people from within their local community, or from those within the wider farming community who live alongside the species in question (Wood et al., 2014). This presents an opportunity for conservation organisations to involve interested farmers in conservation planning and account for experience, to ensure that farmers feel represented in the process.

Reintroducing the wildcat

Support or opposition for the reintroduction of species can be shaped as much by environmental or political ideology, and our environmental value orientation, as by knowledge of the species (Hiroyasu et al., 2019). In the case of the wildcat, a lack of information, particularly regarding the size and diet of wildcats, was a factor for those who opposed and supported a reintroduction. This is not surprising for a species that has been absent for over 150 years and suggests that the cultural salience of wildcats in these landscapes is low. Regional extinction can ultimately lead to the loss of a species from the collective memory of a society (Jarić et al., 2022). While wildcats have not reached that status yet, our results are in contrast to Scotland, the last refuge for wildcats in Britain, where 'Scottish' wildcats retain a symbolic and cultural value (Williams Foley, 2022). Moreover, the strength of the association between wildcats and Scotland could have resulted in a dissociation, over generations, from wildcats as a native species to England and Wales. It is likely then, that responses in England and Wales are in many instances based more on underlying values toward the environment and conservation, than on awareness of the species itself. This calls for effective engagement to reintroduce wildcats to the cultural landscape. While the absence of knowledge could mean farmers have fewer preconceived ideas about wildcats, it could also lead to misinformation filling the void. For example, associations between wildcats and unowned (feral) cats which are frequently controlled may lead to conflict due to similarities in their appearance.

Among those who expressed support, benefits such as control of grey squirrels were cited. The absence of grey squirrels from the majority of the wildcat distribution means the extent to which they constitute a food source for wildcats, and the impact of predation by wildcats, are unknown (Apostolico et al., 2016; Lozano et al., 2006). Farmers' motivations for supporting a reintroduction were primarily led by their interest in 'what wildcats can do for us', which also included suppression of fox populations and pest control. Conservation groups, therefore, need to be careful to manage expectations when disseminating information, to ensure that support is not acquired based on misinformation or undue optimism.

Wildcats were generally not perceived by farmers as a threat to livestock. There is no published evidence of wildcats predating on sheep or lambs, nor has it been reported by farmers as an issue within wildcat areas of Scotland, while predation of poultry is thought to be rare. Indeed, the National Farmers Union of Scotland have expressed public support for wildcat reinforcement programmes (BBC, 2023). Awareness of farmer experience in other wildcat areas is likely to be important for the acceptance of the species within farming communities. The need for lethal control, if any conflict did occur, was mentioned by those who expressed some concern over the potential size of wildcat populations. Direct conflict with human interests is a common reason why reintroductions fail (Dando et al., 2022). Ensuring key stakeholders such as farmers are included in the reintroduction process and that concerns are addressed and experience utilised is important to both project outcomes and farmer-conservation relationships.

Where opposition was more fundamental and, in many cases, not related to wildcats, it was founded on the view that reintroductions go against a participant's vision for the landscape. Much of this can be linked to a desire to preserve cultural heritage (Deliège, 2016; Wynne-Jones et al., 2018), in the face of calls to shift practices in part due to biodiversity collapse (Deliège, 2016; Lorimer et al., 2015; Navarro and Pereira, 2012). It may prove challenging to change strongly held views on the landscape, however, engagement could soften opposition simply through building relationships and incorporating both cultural and biological diversity within projects.

The conflation between a species reintroduction for a conservation benefit, such as for the nationally, critically endangered wildcat, and wider rewilding discourses and activities has been highlighted in similar studies (Bavin et al., 2020) as has the potential of rewilding to be harmful in building relationships with affected communities. Rewilding projects and their advocates vary in vision and application (Thomas, 2022) yet rewilding was frequently described as 'anti-farmer'. In Britain, while large carnivore reintroductions are frequently debated/proposed (Bavin et al., 2023; Hawkins et al., 2020; Hetherington, 2006; Johnson and Greenwood, 2020; Nilsen et al., 2007; Ovenden et al., 2019; Smith et al., 2015; Wilson, 2004), none has been reintroduced. Despite this, the perception of, and subsequent hostility towards rewilding in farming communities (Sandom and Wynne-Jones, 2019) is centred on reintroducing wolves and lynx. Ensuring a reintroduction has a clear and specific focus was mentioned as helping attain support, therefore setting out the aims of a wildcat reintroduction project early and being transparent about its scope will likely be helpful in engagement. This is especially true in the Welsh landscape, where rewilding is controversial following the prominent failure of previous rewilding initiatives (Bavin et al., 2020; Jones, 2022; Wynne-Jones et al., 2018).

Our study highlights the importance of understanding the values, expectations and preferences of key stakeholders in facilitating positive engagement with reintroduction projects. Doing so can create a platform for effective consultation and the development of constructive relationships. This is especially true for species where cultural salience is low, such as in the case of the wildcat. Understanding what shapes attitudes in the absence of species connection is critical to predicting support and communicating effectively. While farmers largely did not view the current relationship with conservation organisations as positive, they were willing to engage with conservation practices when 'done right'. This included in-person and localised engagement, which are crucial to establish conservation scientists and organisations as trusted sources of information but need care and time to deliver. The findings from this study are placed in the context of the reintroduction of wildcats, but the results have wider relevance to the conservation of wildlife in any farmed landscape and stakeholder engagement in conservation more generally.

Chapter Four: Cat owners’ perceptions of unowned cats and implications for European wildcat (*Felis silvestris*) restoration

This chapter has been submitted to *Biological Conservation* as:

Dando, T.R., McDonald, R.A., Young, R.P., Carter, S.P., and Crowley, S.L., 2024. Cat owners’ perceptions of unowned cats and implications for European wildcat (*Felis silvestris*) restoration

Abstract

Domestic cats *Felis catus* exist on a spectrum of proximity in their relationships with people, related to variation in the degree of human control over aspects of their lives. Cat owners are key stakeholders in cat management. Their perceptions of cats’ impacts, and human responsibilities for cats, substantially influence discourses around cat management. Understanding cat owners’ perceptions is particularly important in places where domestic cats might interact with European wildcats *Felis silvestris* and where the status of the latter is threatened by hybridisation with domestic cats.

We conducted semi-structured interviews with cat owners living in rural areas of southwest England and southwest Wales, two regions in which European wildcat reintroductions are being contemplated. Interviewees were asked about their perspectives on the impacts of, and responsibility for, owned and unowned domestic cats, and wildcats. We also explored cat owners’ present knowledge of wildcats.

Unowned cats were perceived to have similar welfare needs to owned cats. Predation of wildlife by unowned cats was viewed as beneficial for pest control or necessary for cat welfare, whereas for owned cats predation was considered problematic. Unowned domestic cats and wildcats were both viewed as a threat to owned cats and were perceived as ‘wildlife’ in the context of interactions between cats.

There was a lack of consensus over who is, or should be, responsible for unowned cats. By contrast, there were clear lines of responsibility for owned cats and wildcats, ascribed to cat owners and wildlife conservationists, respectively. We identify the shifting perceptions of unowned cats between domesticated and wild, and the lack of cognisance of wildcats among cat owners, as primary challenges facing unowned cat management and wildcat restoration.

We propose the need for collaboration among a broad group of stakeholders to develop management strategies for unowned cats in the context of wildcat restoration and suggest a focus on benefiting cat welfare. The challenges of domestic cat management provide insight useful to wildlife restoration projects where there is a need to engage and highlight co-benefits with stakeholders who are interested, but not necessarily engaged in conservation issues or practices.

Keywords: European wildcat, *Felis silvestris*, Domestic cats, Feral cats, Reintroduction, Social feasibility, Stakeholder Engagement, Cat management

Introduction

Domestic cats (*Felis catus*) can threaten biodiversity through wildlife predation (Doherty et al., 2016; Loss and Marra, 2017), compete with wild carnivores (Medina et al., 2014), and act as both reservoirs and carriers of infectious diseases (Gerhold and Jessup, 2013; Taetzsch et al., 2018). Domestic cats also present a challenge to the conservation of the European wildcat (*Felis silvestris*), with which they readily interbreed, leading to the introgression of domestic cat DNA into wildcat genomes (Tiesmeyer et al., 2020). These diverse impacts, in combination with the profound relationship between domestic cats and people (Crowley et al., 2020a), make the management of domestic cats a keenly debated and at times divisive topic (Calver et al., 2020; Crawford et al., 2019; Hostetler et al., 2020; Read et al., 2020; Wolf et al., 2019). Cat owners are key stakeholders in these debates, as their ownership practices directly influence the impacts that cats have (Bassett et al., 2020; Cecchetti et al., 2021a; Cecchetti et al., 2021b; Escobar-Aguirre et al., 2019), and their perspectives influence wider discourses on cat management and the policies and actions of welfare organisations (Deak et al., 2019; Loyd and Hernandez, 2012). Understanding cat owners’ perspectives on the impact and management of cats is therefore helpful for effective deliberation, decision-making and conservation practice.

Management of domestic cats and mitigation of their impacts are complicated by their variable relationships with humans. Domestic cats exist on a spectrum of behavioural autonomy and legal regulation: they can be owned, semi-owned or unowned depending on the level of human responsibility and control over their movement, feeding and reproduction (Crowley et al., 2020a). Cats living as owned companion animals experience more control, while unowned (feral) animals are largely uncontrolled. Diversity and overlap exist within and among these types. For example, owned indoor-outdoor cats are fed by people while their movement and reproduction are often curtailed. Similarly, while having no controls placed on their movements, free-ranging cats are often treated as ‘working animals’, including on farms, where they are fed and sometimes neutered (Crowley et al., 2020a).

The ownership status of cats can be dynamic, with owned cats having the potential to stray and become feral, while unowned cats can return to greater human control through rehoming or being kept as working animals on farms (Bradshaw et al., 1999; Crowley et al., 2020a). The dual processes of 'feralisation' and 'domestication' (here indicating a within-lifetime transition into closer relationships with humans) are complex (Henriksen et al., 2018; Bonacic et al., 2019) and affect both the impacts of, and assumed legal and ethical responsibilities for, domestic cats. People's perspectives as to what constitutes different cat 'types' are similarly complex, and can change depending on the space the animal inhabits and its behaviours (Van Patter and Hovorka, 2018). Moreover, people's perspectives of cats can vary among cat generations due to hybridisation (for example, European wildcat-domestic cat hybrids) and human interventions, including cat ownership practices, cat management activities and conservation engagement (Fredriksen, 2016; Gering et al., 2019; Rodríguez-Rodríguez et al., 2022).

The processes of feralisation, domestication and hybridisation place cats in a liminal biopolitical space between nature/culture and domesticated/wild (Fredriksen, 2016; Holm, 2020; Palmer and Thomas, 2023; Van Patter and Hovorka, 2018). The distinctions made by people between an owned cat and an unowned cat, or a hybrid cat and a wildcat, have material consequences; each cat's typology, and those making a judgement of this, subject it to shifting combinations of perceptions, social constructs, legal protections and decision-making processes (Fredriksen, 2016; Holm, 2020; Srinivasan, 2013; Trouwborst et al., 2020; Trouwborst and Somsen, 2020). Furthermore, definitions of welfare, ethics, and status will change depending on the human actor, place, time and circumstance, meaning all domestic cats could be seen as companions or pests (Crowley et al., 2020a), and subjected to care or persecution, lethal or non-lethal control, or all of the above, at some stage, irrespective of the cat's individual actions (Holm, 2020; Srinivasan, 2013).

Accordingly, owner attitudes towards responsibility for the impacts cats might have and the care they might need are diverse and vary between cat behaviours and types (Habacher et al., 2010; Toukhsati et al., 2007; Wald et al., 2013). On the issue of hunting by cats, owners who were aware of their cat's impacts were more likely to feel responsible for managing behaviours (Crowley et al., 2020b, 2022). In contrast, others believe they have no responsibility to manage hunting as they feel they cannot stop it, or view it as natural rather than problematic (Crowley et al., 2020b). Similar divisions are identifiable in relation to outdoor access, with some owners perceiving unimpeded access as essential, while others feel it is responsible to manage their cats' access, either to enhance cat welfare or to mitigate negative impacts on wildlife (Crowley et al., 2020b, 2022). Previous research indicates that owners attribute responsibility for the care of unowned cats, in particular around neutering, to communities, councils and charities (McDonald et al., 2018; Slater et al., 2008; Vasileva and McCulloch, 2023). Conversely, however, some people feel responsible for providing caregiving to unowned cats, including feeding and neutering due to concerns for unowned cat welfare (Finkler et al., 2011; Toukhsati et al., 2007; Zito et al., 2015). Reconciling this diversity among cat owners with successful reduction of impacts is a key challenge when designing effective cat management strategies.

Management of unowned cats is conducted through lethal and non-lethal means (Cecchetti et al., 2021b; Kennedy et al., 2020). Understanding the effectiveness of different management approaches and stakeholder perspectives of these is important in achieving successful outcomes (Ahn, 2022; Deak et al., 2019). Trap-neuter-vaccinate release (TNVRe), trap-neuter-vaccinate-rehome (TNVRh), and lethal control are common tools in the management of unowned cats, but public support/tolerance for each varies significantly between countries and different demographic groups (Hall et al., 2016a). Proximity to natural or semi-natural areas also influences the extent to which people perceive cat management to be needed (Bassett et al., 2020). Each method has ethical implications, including the ethics of releasing cats, for example after neutering, and thus allowing them to continue hunting wildlife (Crawford et al., 2019; Johnston, 2021). The perceptions of cat owners are also important to discussions of cat management (Crowley et al., 2020a; Elliott et al., 2019). Owned cat populations can have negative environmental impacts and are a source of unowned feral cats, with individuals often transitioning from owned to 'stray' or feral (Bradshaw et al., 1999). This sometimes rapid transition can be caused by permissive ownership practices which do not control a cat's movement or reproduction and may be an active decision by the owners to relinquish control over the cat, or indeed the cat forsaking its owners (Horwitz and Pike, 2016).

The spread of domestic cat genes by their hybridisation with wildcat species is a major conservation concern and a significant management challenge. While the closest wild ancestor of the domestic cat is the African wildcat (*Felis lybica*) (Driscoll et al., 2007; Kitchener et al., 2017), hybridisation is primarily an issue for the European wildcat (Le Roux et al., 2015; Tiesmeyer et al., 2020). This species is present throughout much of Europe, where it is locally threatened by hybridisation and disease transmission from domestic cats, as well as by road mortality, human disturbance and intensive agriculture (Bastianelli et al., 2021; Daniels et al., 1999; Mattucci et al., 2016; Ruiz-Villar et al., 2023; Tiesmeyer et al., 2020; Unterköfler et al., 2022). Hybridisation is most prominent in Scotland, where the last remaining wild wildcat population in Britain has been described as 'functionally extinct' as a consequence (Breitenmoser et al., 2019). To counter this situation, wildcat reinforcements are being undertaken in Scotland, and reintroductions have been proposed in Wales and England (Breitenmoser et al., 2019). The importance of managing domestic cats (including owned, semi-owned and unowned animals) and engaging with cat owners have been identified as fundamental to the long-term success of wildcat conservation strategies in Britain (Campbell et al., 2023b; Littlewood et al., 2014). While some engagement has been undertaken as part of conservation initiatives within wildcat 'priority areas' of Scotland (Bacon, 2017; Littlewood et al., 2014), this is not the case in England and Wales, where wildcats have been absent for ~150 years (Langley and Yalden, 1977). With reintroduction efforts proposed in these landscapes, an exploration into current cat owner perspectives is required to inform the development of an engagement strategy and as part of the assessment of the social feasibility of a reintroduction.

To explore cat management and owners' perspectives, we conducted semi-structured interviews with cat owners in rural areas of England and Wales. The interviews considered variations among owners' perceptions of impact and responsibility concerning cat life history and owners' current awareness of European wildcats and their attitudes to a proposed reintroduction.

Methods

We conducted semi-structured interviews with cat owners. We aimed to examine how owners' perceptions of impacts and responsibility for cats vary according to a broad classification of cat 'types'. To enable discussion, we adopted a simple classification of domestic cats as owned or unowned, based on Crowley et al (2020a) in relation to provisioning, reproduction and movement. We also used wildcat as a further type, referring to the European wildcat. Interviewees were informed how owned and unowned cats were defined by this study as a preface to the interview questions. While we used the term unowned cats, interviewees frequently used the synonym ‘feral cat’; similarly, domestic cat was used interchangeably with owned cat, and we retain these terms when participants used them. Wildcats were not defined for participants, as we were interested in determining cat owners’ existing knowledge of this species.

Participants were recruited and interviews were conducted between November 2020 and April 2021. We targeted areas under evaluation as potential regions for wildcat reintroductions (MacPherson et al., 2020) in southwest England and southwest Wales. Recruitment occurred primarily online, by advertising through a diverse selection of ‘cat groups’ through social media and via snowball sampling. Four main online groups were targeted: interest groups, rescue centres, welfare groups and farm/feral cat groups. This approach to recruitment means participants were likely to be more than usually engaged with the issues discussed. Consequently, our sample is likely to be biased toward those interested in, and who feel most strongly about, cat welfare and management, rather than representative of the broader population of cat owners. Such engaged cat owners are especially important to these discussions due to their influence on and contribution to the discourse about cats and their management. We were seeking a diverse rather than representative sample, and towards the end of the recruitment period we purposively recruited those who had rehomed unowned animals as outdoor-only cats, recognising the potential significance of this group in terms of these animals’ ecological interactions. Recruitment advertisements included a summary of the interview topics and geographical target, as well as details of the interview process itself. Additional participants were recruited via word of mouth. We focused on rural areas as defined by Rural Urban Classification for England and Wales (Department for Environment, Food & Rural Affairs et al., 2016). Cat owners in rural areas are exposed to the interface between owned and unowned cats and would be for wildcats in the event of a reintroduction.

We interviewed twenty-five people, of whom 23 (92%) were female. This sampling bias has been seen in similar studies of cat owners (e.g. Wald, Jacobson and Levy, 2013; Hall *et al.*, 2016a; Crowley et al., 2019). While there are more female cat owners in the United Kingdom (58%: PDSA, 2017) such a strong gender bias in response to recruitment in this study and others indicate the need for future studies to seek alternative approaches to recruitment that incentivise male cat owners to participate. Similarly, our sample contains a large skew (94%) toward owners who neuter their cats, however, this is broadly consistent with an estimated average neutering rate in the UK of 89% (PDSA, 2022).

TD conducted all interviews, following a semi-structured schedule. Interviews were conducted using both video and telephone calls due to restrictions during the Covid-19 pandemic. All interviews were audio-recorded and were transcribed verbatim. Participants read an information sheet explaining the topic of the study and were sent consent forms via email which were signed and returned before their interview. Participants were informed that they had the right to withdraw at any time. The study received ethical approval from the University of Exeter's CLES Penryn Ethics Committee (eCORN003107).

Transcripts were analysed using NVivo (v12). Following an initial read-through, text was thematically coded in three stages by a single coder (TD) to identify the most important themes (not necessarily the most common responses) that emerged concerning each cat classification. First, transcripts were coded by grouping responses by cat classification. The resulting grouped responses were coded into themes relevant to perceptions of (a) impacts of each cat type; (b) responsibility for each cat type; (c) management of each cat type and (d) wildcats and their reintroduction. The content of each theme was then sub-coded to combine related perspectives within each theme. Data were interpreted in relation to the existing literature and wider discussion with owners. Numeric coded identifiers are used here to protect participants' identities.

Results

This section is organised in relation to four key topics from our interviews. All highlight the importance of cat type in determining participants' perceptions and attitudes towards cat behaviours, impacts, and appropriate management. First, we consider participants' perspectives of the impacts of owned cats, unowned cats, and wildcats upon wildlife, other cats, and upon people. Second, we highlight the variable importance of cat welfare as a consideration. Third, we identify how participants attribute management responsibility to different people and/or organisations depending on cat type. Finally, we focus on cat owners' views on the prospect of a wildcat reintroduction to their area.

Impacts and Interactions

Impact on wildlife

The impact of unowned cats and wildcats on wildlife, specifically arising from predation, was viewed by some participants as beneficial, however, others expressed concerns over their impacts (Figure 4.1). A key component was differing views on cat predation on ‘wanted’ (e.g. birds) vs ‘unwanted’ wildlife or pests (e.g. rabbits and rodents). For unowned cats and wildcats, beneficial control of ‘pests’ was identified (“*These guys are mainly catching rabbits and we have plenty of rabbits around... we need to keep them down; they [rabbits] do a lot of bad down at the farm*” [Participant 05]). In contrast, participants expressed concern over owned cats’ impacts on birds and the suffering they cause, due to them enjoying having birds around (“*I’m not concerned about the mice he’s catching, but the birds, I don’t like it when he brings them in, because I like birds*” [18]). As with unowned cats and wildcats, in the circumstances where participants found predation to be positive, it involved rodents (“*the main reason for having outdoor cats is because we have had rats and mice problems in the past*” [17]); typically, the latter were participants with outdoor-only cats.

While it was acknowledged that unowned cats could pose a threat to wildlife populations, this was often caveated with understanding and sympathy (“*I just feel bad for them, but I know it’s bad for the environment because they kill lots*” [14]). In comparison, concerns over the impact of owned cats on sensitive wildlife were usually unequivocal (“*We’ve got farm fields behind and the bird sanctuary down the bottom. If the cat was to take some of the small baby birds… What's the impact of that? What are the knock-on consequences? It can be quite huge*” [06]). Wildcats, by contrast, were generally described in positive terms regarding their impact on wildlife populations, due to an understanding of wildcats being native and therefore part of British ecosystems ("*the ecology of that place is geared to that animal being there*" [02]).

A few participants felt that owned cats posed no threat at all to wildlife populations ("*It's a complete assumption and it is overstated*" [09]). Similarly, the relatively great availability of space in rural areas was balanced with the number of unowned cats by one participant, who concluded that the impact of unowned cats would not be noticeable ("*There's a lot of space around, so if there were 10 feral cats living near me you probably wouldn't notice that much*" [15]). Some participants acknowledged that they did not know enough to assess the impact of wildcats on wildlife populations but suggested that research is needed before any reintroduction into rural landscapes ("*I don't know but I would hope someone has done good research to figure out the impact that they would have on the environment*" [12]).

Impact on other cats

Participants described conflict between cats as an impact associated with wildcats and unowned cats more than owned cats (Figure 4.1). Owned cats were typically framed as the subject of concern and potential victims, while unowned and wildcats presented a threat. Unowned cats were often characterised as being "vicious" and "dominating", compared with "calm" owned cats ("*domestic cats tend to be a lot calmer and would probably back away if he engaged with them... they tend to be a nicer character than some of the feral cats, who tend to be quite dominating*" [06]). Similarly, for wildcats, some were fearful that their cats would be attacked ("*I would be a bit nervous maybe, about my cat getting attacked by them*" [18]). However, others thought owned cats would likely avoid conflict with a wildcat ("*people say that it might cause an issue to domestic cats. I don't think it would unless you've got a very stupid domestic cat*" [08]).

Participants identified territorial behaviours of owned cats as a possible cause of conflict with other owned cats as well as unowned animals ("*I think cats don't like other creatures being in their territory... one of the Savannahs [hybrid of domestic cat and serval] has taken to marking territory which I think is related to that feral cat*" [01]). There were also concerns that unowned cats could spread disease ("*I don't know where this cat hangs out but he's often around here. It is* *a concern if they might be carrying a disease*" [01]). Finally, a small number of participants described hybridisation occurring between cat types. This was framed by some as being a possible incentive for more people to neuter their cats if they were aware of the issue ("*If you've got the risk of your cat breeding with a wildcat, it might pressure people into thinking, well I might neuter him*" [08]).

Impact on people

Concerns over any negative impact of owned, unowned and wildcats on people were rare for all cat types (Figure 4.1). Owners instead often described owned cats as having a positive impact on people, specifically their owners ("*I've had issues through the last 10-15 years because of my ill health, and a cat has been very helpful for me personally*" [10]). The only negative impact of owned cats on people cited involved damage to gardens ("*It is frustrating when a cat comes along and messes up your lawn*" [13]).

Negative impacts on people were more frequently described for wildcats, with participants concerned that wildcats may impact people's livelihoods, most notably, sheep farmers, pig farmers and poultry farmers, through predation of livestock ("*farmers would probably be the main kind of ones that would be concerned because, obviously, putting lambs out, I'm guessing that they're capable of taking the lamb or a piglet*" [03]); the latter was also reflected in opinions of unowned cats. Finally, one participant expressed concerns for dog walkers that their pets might be targeted ("*When people have tiny little dogs if food is scarce, and a little dog is running around, it is going to get eaten by a big Scottish wildcat*" [09]).

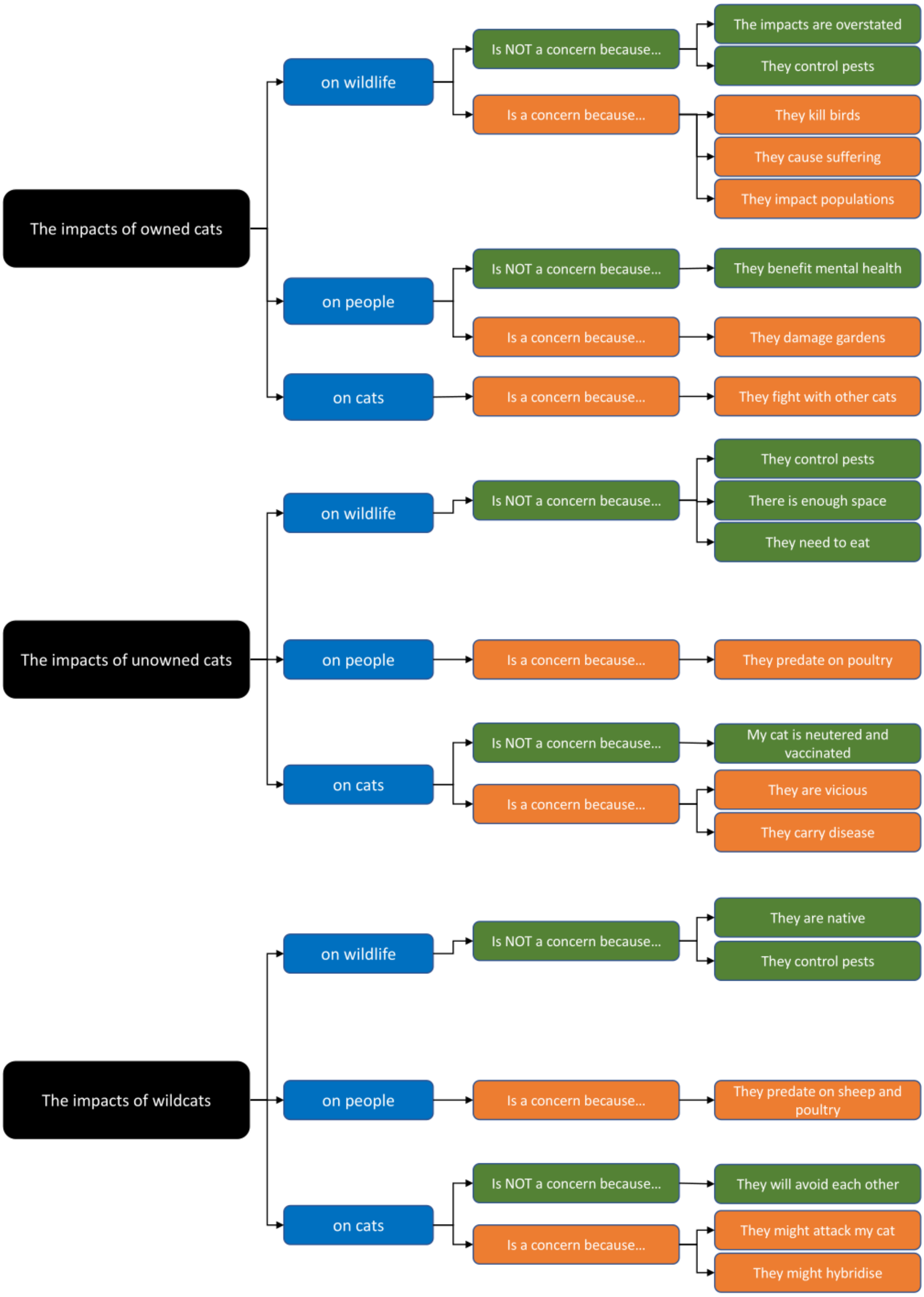

Figure 4.1 Schematic showing range of owners' perspectives on the impact of owned, unowned and wildcats. See text for details and examples

Cat Welfare

Welfare was discussed as a concern for owned and unowned cats, but was largely absent from conversations about wildcats. Participants primarily focused on concerns around road mortality for owned cats ("*The main issue for cats going out, without a doubt, is road traffic accidents*" [21]). In many cases, participants identified issues around high populations of cats as being an urban issue rather than one they faced as rural cat owners ("*There's obviously issues where you've got a suburban environment or urban and there are lots of cats in the area, the cats can get stressed because they're territorial*" [21]).

Welfare was the most frequently discussed issue for unowned cats. Responses coalesced around concerns over their health, linked to a perception that they are more likely to suffer from a disease, poor living conditions and both inbreeding and overbreeding compared to owned or wildcats ("*if you leave them alone, they will just breed and breed and get diseased. If you ever see any feral populations, they always look scraggly and have eye and nose problems, they aren't healthy, because at the end of the day, they're domestic animals, not wild animals*" [19]).

The ability of cats to adapt to outdoor environments was a common point of divergence and contradiction in respondents' views on owned and unowned cats. Unowned cats were described as being "*survivors*", "*resilient*" and "*wild*" while also being described as "*not suited to wild environments*" and "*needing care*". These characterisations were not always mutually exclusive in responses by individual participants, for example, "*Feral cats are just great adapters and true survivors"* [20] and *"Ultimately there needs to be management [of unowned cats] to protect them from an unforgiving environment, simply for the welfare of those animals"* [20]. The contrast in views over whether unowned cats are adapted to the wild or are dependent upon people was emphasised when discussing the management of unowned cats; owners were divided over the welfare outcomes of releasing or rehoming unowned cats during TNVR programmes. Lethal forms of management were not considered acceptable by any interviewee. Participants described indoor-only cats as "*bored*" with "*odd behavioural traits*" and "*built to be outside*" while others refused to let their cats out in the rain or cold and felt they were dependent on attentive human care. A key point of difference was that the welfare of owned cats was positively linked to access to veterinary care, while owners felt this was lacking for unowned cats. Wildcats were generally not perceived to have any welfare issues. This was explained due to them as being naturally adapted to a wild life ("*Wildcats are built for the environment they're in, which is different from ferals, that's their world and they know how to survive*" [25]) Owners also focused on the physical appearance of the animal as a guide for its state of welfare ("*I think it would bother me if they didn't look well and they didn't look looked after*" [15]).

Responsibility

Responsibility for the behaviour and impacts of owned cats was split by the respondents into four parties (Table 4.1), cat owners, cats, society, and cat charities:

Cat owners identified themselves as being responsible for all aspects of an owned cat's health, and sometimes any impact they might have inside and outside of the home ("*I am responsible for their health, their wellbeing. And then also I want to make sure they're not impacting on wildlife, like some of the birds, particularly*" [05]).

Cats were determined by a minority of participants to be responsible for their own actions outside of the home, such as predation and social disturbance, commonly stating that they (owners) cannot control what their cat does ("*When they're outside you can't really stop what they're doing… it's kind of a wild animal. Because you can't control that unless you're going to fence them in or have electric fences like they have in America, which seem really horrible*" [08]). This did not, however, extend to mitigating the risk of road mortality, which participants thought was the responsibility of cat owners.

Society (meaning everyone) was considered responsible for being aware of, and reporting, owned animals or cats of unknown status they encounter if they are in bad health.

Cat charities were deemed responsible for ensuring owners are "*educated*" and able to adequately care for their cats.

Designated responsibility for unowned cats was split among seven parties (Table 4.1): cat charities, government and local authorities, farmers and landowners, society, cat owners, wildlife conservation organisations, and a coalition of some or all of these parties.

Most commonly, *cat charities* were cited as being responsible for aspects of unowned cat health, breeding, and freedom. This was frequently linked to TNVRe or TNVRh programmes ("*I think we should help them. I donate to cat charities every month and they do lots of trapping and provide health checks, neutering that sort of thing, so they would be my first port of call*" [14]).

Government and local authorities were identified as being responsible for aspects of freedom, breeding and identification of unowned cats, the latter referring to having a more robust and regulated microchipping system, to improve accountability for them ("*a government programme would be quite useful to have if they were to introduce (wild)cats, if you had to have microchips it would be a lot easier to identify a feral cat*" [8]).

Farmers and landowners were deemed responsible for unowned animals on their land in terms of maintaining their general health and limiting breeding. Participants argued that if farmers gain a benefit from the cats in the form of free pest control, then they should be required to provide a basic level of care ("*If an animal is of use to you as a farmer, you know if you're prepared to take those services and keeping your rodent population down or whatever. I think on a human level you've got some responsibility to them*" [23]).

Society, as with owned cats, was thought to have some responsibility for unowned cats, with participants suggesting that everyone should be vigilant in reporting unowned cats in poor health, or potential feral and stray animals to vets, community groups and charities ("*I think they are society's responsibility really, nobody owns them but they are suffering and if there are feral cats in a community, the people living there should be trying to do something about it, rehome or whatever, but so many people just don't care*" [25]).

A *coalition of stakeholders*, including landowners, charities and local government was also suggested as being needed to reduce numbers and provide better welfare outcomes for unowned cats ("*I suppose it's got to be a mix of landowners and charities and the local council working together to ensure that populations are managed and controlled*" [17]).

Finally, *cat owners* and *wildlife organisations* were each mentioned once. Cat owners might reduce the impact of owned cats becoming feral by microchipping and neutering them, while it was suggested that wildlife organisations could intervene around the issue of unowned cat predation on wildlife, although no detail was given as to how.

For wildcats, responsibility was predominantly attributed to the *wildlife organisations* interested in reintroducing them (Table 4.1), referring to research to ensure there is enough food for them and that any risks to cats and other rural industries are mitigated ("*I would have thought it would be up to the wildlife groups releasing them to make sure their needs were met and whatever research was needed to decide that was done beforehand*" [25])

A few participants mentioned *cat owners* as being indirectly responsible for wildcats by reducing the potential for conflict and hybridisation, for example, by adapting their cat's access to the outdoors at certain times.

Table 4.1 Described responsible parties for the care and management of owned, unowned and wildcats. The colour scale reflects the frequency at which interviewees cited each stakeholder, with darker colours being more frequent

	Domestic cats		Wildcats
Stakeholder	Owned	Unowned	
Cat charities			
Cat owners			
Cats			
Coalition of stakeholders			
Government authorities			
Landowners/Farmers			
Society			
Wildlife organisations			

European wildcat reintroduction

Interviewees primarily supported a wildcat reintroduction. Interviewees highlighted that wildcats are a "native animal", and that wildcats "*belong*" in the landscape (Figure 4.2) ("*I think If there's a wildcat reintroduction it's only putting back what should be there in the first place. And that's a different scenario to a domestic cat, which is not necessarily supposed to be in the area*" [02]).

Others expressed support for reintroduction because of the wildcat's current poor conservation status, or simply because they loved cats ("*I am a cat person and I love cats, I would love to see them in this country*" [13]).

Control of other species such as rabbits was also mentioned as a potential benefit of wildcat reintroduction, while wildcats were viewed by one owner as a replacement for rehomed feral cats, proposing that their reintroduction would result in people having less need to control the rodent population ("*I would have thought of them [wildcats] as a replacement and therefore reducing the need for people to keep their own even domestic or farm or feral cats*" [22]).

A small number of participants expressed support for a reintroduction with caveats or conditions. For example, they liked the idea but did not feel they knew enough about wildcats to offer an opinion (Figure 4.2). Others had concerns about the potential impact of wildcats on farmers through predation of livestock, as well as the availability of suitable habitats ("*I suppose in Scotland there's so much kind of wilderness, but we don't have that amount of wilderness down here in England, do we?*" [09]).

The few owners who opposed reintroduction did so due to concerns over conflicts with their cats, however, two of these supported a wildcat reintroduction provided it didn't happen near them (Figure 4.2) ("*Oh my god. No, I wouldn't be very happy, maybe somewhere like all them moors down between St Ives and Land's end, but not here*" [09]).

The support expressed by participants is caveated by a recognised lack of knowledge about wildcats among many owners. Participants frequently stated that they didn't know much about wildcats, or responded to questions with clarifying questions of their own, around interactions ("*I don't know if wildcats interact with domestic cats or whether they just don't*" [22]), diet ("*I don't think a wildcat could take a deer could it?*" [03]) as well as size and type ("*I haven't* *actually heard about wildcats, I've heard about lynx, is that what you mean?*" [08]) (Figure 4.2). Hybridisation was raised by only four owners, suggesting that this, the primary issue of conservation concern for wildcats, was not common knowledge.

Finally, owners linked wildcat reintroduction to potential changes in their behaviour toward outdoor access, with owners expressing an inclination to keep their cats in at night if wildcats were in the area, to reduce the risk of conflict. Owners were split on whether a wildcat reintroduction would lead to a greater uptake in neutering, with some suggesting owners would be more likely to neuter if wildcats were present, provided it was accessible ("*most people, I should think, would be onto neutering if there is a good program or cheap enough to do it because some of the vets around here are so expensive. I think for people who have got low incomes, it could be a real problem*" [05]). Others felt that wildcat reintroduction would not make owners more likely to neuter, perceiving that most who don't neuter cats avoid this because of set beliefs ("*I think if you don't think it's important now you're probably not going to change your mind, its sort of the ethical reason of, 'I believe cats should live wild'*" [17]).

Figure 4.2 Summary of cat owner responses to the prospect of reintroduction of European wildcats to their area. Divided into four groups: supportive statements, conditional support, opposition statements and questions

Discussion

How cats lived and the proximity of their relationships with people, influenced how cat owners perceived both cat impacts and human responsibility towards cats. We found disparities in participants' perceptions towards potential impacts, with owned cats perceived to have more negative impacts upon wildlife than any other cats. The converse was true for wildcats, which were thought to be beneficial to wildlife. Unowned cats, meanwhile, were thought to have both negative and positive impacts on wildlife; however, owners accommodated the negative impacts by recognising their need to survive. This contrasts with participants' perspectives on conflict between cats, with the perceived 'wild' behaviours of unowned cats and wildcats viewed negatively and not acceptable in the context of interactions with owners' cats.

Participant expectations of unowned cats' welfare, specifically in the context of cat management and the merits of TNVRe and TNVRh varied in alignment between that of owned cats and wildcats depending on the participant. Moreover, so did participant perceptions of their impact on the environment and other cats. Unowned cats, therefore, exist in a liminal state in cat owners' perceptions of impact, responsibility, welfare, and consequently, management, connected to both domestic and wild environments but belonging to neither. This diversity of viewpoints is a key challenge in developing management strategies for unowned cats, with associated implications for cat welfare, cat populations and for impacts of cats on wildlife. Furthermore, failing to develop and implement management strategies has negative consequences for wildcat restoration due to the risk of introgression posed by populations of unowned cats.

Cat impact, welfare and responsibility

Our interviews highlight that the impacts of cats on wildlife are perceived differently by cat owners depending on cat 'type'. Predation by owned cats was viewed as anthropogenic and was consequently seen as unnatural (Holm, 2020). Predation of birds in particular was viewed negatively (Foreman-Worsley et al., 2021). In contrast, as an indigenous wild animal, predation by wildcats was considered 'natural', and therefore any impact was also perceived this way. Only in certain circumstances was predation by owned cats viewed as desirable, for example when catching and killing 'vermin' such as rodents (see also Crowley et al., 2019). Predation by unowned cats was thought of as useful when *in the right place*, indicating unowned cats functioning as a 'domestic wildcat'. Furthermore, predation by unowned cats that are hunting to survive was perceived as more acceptable due to the poor welfare conditions in which they were perceived to live. Cat owners in our study are, therefore, shown to prioritise unowned cat welfare over any negative ecological arguments they may have. This underscores the significance that cat owners place on domestic cat welfare, regardless of ownership status.

Concerns for unowned cat welfare and sympathetic attitudes displayed by owners toward unowned cats may be affected by comparisons with their caregiving to owned cats. Both in our study and others, pet owners perceived their actions to create a high welfare environment (Westgarth et al., 2019). In this context, it is understandable that any cats that do not receive similar levels of care, such as unowned animals, are perceived to be suffering in an environment characterised as dangerous and unpredictable. There were, however, differences between those who viewed unowned cats as needing human care to meet a minimum standard of welfare, and those who saw unowned cats as ecologically adapted animals expressing innate wild behaviours, this was especially prominent during discussions around TNVR (release or rehome). In this respect, unowned cats were treated more like wildcats, even though the latter were not the subject of welfare concern. This duality has previously been identified in conversations around owned cats (Crowley et al., 2020a), but the prevalence of a similar division in discussions on unowned cats has a direct bearing on debates as to what is considered humane cat management. Effective management of unowned cats requires public buy-in, meaning any approach needs to meet the expectations of the society in which it is taking place (Deak et al., 2019). The apparently divided view of the wildness and the welfare of unowned cats suggests a cat ownership community in the UK that is united behind the notion that unowned cats need to be managed but divided by the welfare and ethical implications of different approaches to achieve this.

Unowned cats and wildcats were more likely to be described in threatening terms than owned cats. This characterisation is perhaps expected as unowned cats are often viewed as removed from 'civilised communities', while 'wildness' is central to the wildcat identity (Fredriksen, 2016; Slater, 2007). While owners described, often in graphic terms, their cat's predatory behaviours (in a way they did not for unowned cats and wildcats), when describing antagonistic interactions between cat types, owned cats were perceived as victims, while unowned and wildcats were framed as villains. The impact of unowned cats in non-human spheres is, in certain circumstances, accepted, due to their welfare needs being closely aligned with those of owned cats. When it comes to interactions between owned cats and unowned cats or wildcats, however, unowned cats are largely treated as wild animals (Holm, 2020), while not ascribed any sense of belonging to either domestic or wild settings.

On human responsibility for cats, we found that unowned cats diverged in owner perceptions from both owned cats and wildcats. The latter two were associated with clear lines of responsibility, albeit with some discussion as to where the responsibility of owners extends regarding roaming and predation of owned cats (as seen elsewhere: (Crowley et al., 2019). However, there was little consensus among owners as to where responsibility for unowned cats lies. We identify two general perspectives, the first of which focuses on responsibility for management initiatives, legislation, and enforcement of regulatory schemes such as microchipping and neutering. In this view, responsibility lies with external authorities such as local governments, cat charities, and wildlife groups, and this broadly reflects findings elsewhere (Gates et al., 2019). The second perspective focuses more on responsibility for unowned cat welfare, identifying those individuals directly interacting with unowned cats – such as farmers, landowners, and cat owners – as responsible. Very few owners expressed the need for collaboration between these sets of actors, and cat owners rarely identified themselves as a responsible group, instead attributing responsibility to others, indicating they largely see unowned cats as outside their sphere of influence. This further highlights the material consequences of the conceptual liminal space unowned cats inhabit; while pet cats are owned and wildcats are conserved, unowned cats have no formal, or broadly accepted, lines of responsibility, hindering the application of collective management strategies at a landscape or societal scale. Instead, responsibility is often self-appointed at a local level, including by charitable organisations (Natoli et al., 2019). To this end, greater awareness of the role that cat owners can play in preventing ferality through responsible ownership should be highlighted in attempts to manage unowned cat populations.

Perspectives on the European Wildcat

Cat owners in our sample knew very little about wildcats. Wildcats were perceived to belong in the English and Welsh landscape, with owners valuing their rarity, and potential role in replacing unowned cats in controlling ‘pest’ species such as rabbits. We did not, however, find the wider range of values attributed to wildcats in assessments of stakeholders in Scotland (Williams Foley, 2022). Notably, there was no reference to cultural, symbolic, or economic value. This is not surprising for a species that has been absent from our survey areas for around 150 years (Jarić et al., 2022), but indicates the need to establish a connection between cat owners and wildcats if owners are to play some role in wildcat restoration or associated management of other cats.

Wildcats were considered by some to be a threat to domestic animals, including the safety of owned cats. This led some owners to support a reintroduction in principle, but not want it to occur near them (Scott et al., 2016; von Essen and Allen, 2020). Domestic cats are rarely found in dietary analysis of wildcats and when they are it is likely as a result of carrion (Biró et al., 2005). The impact of wildcats on livestock (specifically, sheep, pigs and poultry) was perceived to be a concern. While there is little evidence to support the suggestion that pigs and sheep are killed by wildcats (Lozano et al., 2006), poultry has been found, albeit rarely, and less so than for domestic cats (Biró et al., 2005). Wildcats are often depicted as fierce and untameable, potentially increasing the perceived behavioural separation between owned and unowned domestic cats and wildcats (Fredriksen, 2016; Wrigley, 2020). Such representations may be filling gaps in scientific knowledge or practical experience of behaviour and diet (Baker et al., 2020; Lescureux et al., 2011).

Cat owners overwhelmingly deemed wildlife organisations responsible for any detrimental impacts of wildcats if they are reintroduced. The sense that those responsible for a reintroduced species' presence should also be responsible for its actions, at least until the species’ presence has become normalised, has also been documented for reintroduced Eurasian beaver (*Castor fiber*) (Auster et al., 2021b). However, local communities can also become attached to and protective of reintroduced species, such as in the case of the beaver, where some residents took responsibility for the reintroduced animals into their own hands and resisted planned government interventions which sought to recapture them (Crowley et al., 2017a). Long-term commitments by project initiators are important to reintroduction success, but taking responsibility in perpetuity may not be viable for wildlife groups (Dando et al., 2022). Nonetheless, accepting responsibility for an agreed period may be needed to minimise conflict risk and facilitate engagement with key stakeholders until the species' presence is normalised (or protected by the communities themselves) and management processes are established (Auster et al., 2021b).

A small number of participants deemed cat owners to have some responsibility for wildcat welfare, particularly concerning hybridisation between domestic cats and wildcats. This is in contrast to research in wildcat areas of Scotland which found high awareness and strong perceptions of responsibility among cat owners (Bacon, 2017). Neutering programmes targeting owned and unowned cats would likely be a central point to any cat owner engagement on the issue of hybridisation (Campbell et al., 2023b; Kilshaw et al., 2008). However, perceptions of the extent to which a wildcat reintroduction could catalyze these practices were mixed. Despite hybridisation not being a prominent issue in our interviews, we did find that neutering owned and unowned cats was a management priority for our participants, suggesting that despite different motives both conservationists and cat owners support some of the same management strategies (Crowley et al., 2022).

Implications for reintroduction initiatives

Cat owners represent a community that may be natural supporters of wildcat conservation. Collaboration between cat owners and cat welfare groups with wildcat conservation projects will be key to the management of both owned and unowned cats. There is a need to address areas where division exists (e.g. concerns over conflict between wildcats and owned cats), as well as identify areas of consensus, (e.g. the benefits of neutering for wildcats and unowned cat welfare), in establishing meaningful collaboration (Dando et al., 2022). This process is not simply about the meeting of goals for conservation and cat welfare, but also working in relation to the diversity of perspectives that exist among cat owners, especially towards unowned cat behaviour and management. For a reintroduction to be successful, agreement on and a strategy for managing unowned cats is of fundamental importance. The liminal place occupied by unowned cats indicates that conservationists planning wildcat reintroductions need to work with a wide range of stakeholders in the approach and implementation of domestic cat management as a key means of realising their conservation objectives.

Chapter Five: Selecting areas for the reintroduction of European wildcats (*Felis silvestris*) in England and Wales

Abstract

Anthropogenic pressures have caused the decline and extinction of species globally. Conservation translocations including reintroductions seek to restore species but require suitable connected habitat to be successful. We evaluate habitat connectivity to inform the selection of release areas and sites for European wildcat reintroductions in Britain. Using habitat models, we mapped habitat networks in three regions – North Wales, West Wales, South West England. West Wales showed the greatest connectivity among woodland patches, with the largest connected cluster of patches and highest quality linkages. When evaluating potential release sites, an analytical hierarchy process ranked candidate sites in West Wales as most favourable, with lower densities of primary roads and domestic cats, and better sites regarding size and connectivity. While South West England retains large clusters of woodland patches able to sustain a viable population, risks from fragmentation along road networks, and potential human conflicts around release sites appear greater. North Wales is deemed the least suitable overall. Our spatial analysis indicates West Wales provides the most intact, connected woodland networks to support recovering wildcat populations in Britain. More generally, this work demonstrates how a variety of analytical approaches can be used together to improve our understanding and decision-making toward reintroduction planning and feasibility.

Introduction

Anthropogenic pressures are a primary cause of biodiversity loss and species decline (Barnosky et al., 2011; Faurby and Svenning, 2015; Andermann et al., 2020). The spatial extent of anthropogenic activity throughout global ecosystems requires that species restoration efforts occur predominantly in human-modified landscapes (Sanderson et al., 2002; DeClerck et al., 2010; Gardner et al., 2010; Morales-González et al., 2020). A major barrier to species restoration in such landscapes is the reduced availability, quality, and connectedness of suitable habitats. Human activities such as agriculture (Mony et al., 2018), forestry (Liu et al., 2020) or infrastructure development (Liu et al., 2020) can cause loss and fragmentation of habitat, reducing the movement of individuals between populations. This can lead to loss of genetic diversity, and localised extirpations (Keyghobadi, 2007). The impacts of fragmentation can also facilitate negative interactions between wildlife, people, and their domestic animals (Sharma et al., 2020), and/or force species into higher-risk environments, adversely affected by human settlements or roads (Bar-Massada et al., 2014; Bloomfield et al., 2020; Lesbarrères and Fahrig, 2012)

Increasing habitat connectivity is a critical part of conservation. Identifying and improving or restoring important pathways and corridors facilitates the recolonisation or reintroduction of species (Dzialak et al., 2005; Blazquez-Cabrera et al., 2019; Mariela et al., 2020) and contributes to mitigating human-induced biodiversity declines (Taylor et al., 1993; Crooks et al., 2011). Protected areas are often proposed as a means of limiting the negative influence of anthropogenic activity on landscapes, including through safeguarding, and restoring patches of habitat and connectivity (Keeneleyside et al., 2012; Convention on Biological Diversity (CBD), 2021). Yet, within human-modified landscapes, protected areas often do not provide effective protection for habitat of sufficient quantity or quality to support conservation, with protected areas often too isolated to facilitate connectivity (Di Minin and Toivonen, 2015; Shafer, 2015; Schulze et al., 2018; Starnes et al., 2021).

Species reintroductions are a form of conservation translocation, restoring a species to an area within its indigenous range from which it has disappeared (IUCN/SSC, 2013). Reintroductions are increasingly common interventions but are often characterised as high-risk (Stadtmann and Seddon, 2020). They require practitioners to address a mix of social, ecological, and economic pressures (Seddon and Armstrong, 2016; Dando et al., 2022). Moreover, reintroductions require the identification and assessment of release areas, which are of sufficient size to sustain a viable population, and which meet the species' biotic and abiotic needs for all seasons and life stages (IUCN 2013). Within release areas, practitioners also need to select release sites, smaller areas where animals are released from, and which facilitate dispersal into the wider release area, while also meeting the needs of the species and practical needs of implementing a successful release (IUCN 2013). Inadequate habitat within release sites and areas is a leading cause of reintroduction failure (Osborne and Seddon, 2012; Bubac et al., 2019; Stadtmann and Seddon, 2020). Appropriate selection of release sites and release areas is fundamental to reducing ecological risks associated with reintroductions (Stadtmann and Seddon, 2020)

The evaluation of suitable landscapes for reintroduction requires an assessment of social, ecological, and economic factors at varying spatial scales and with a range of methodologies. For example, GIS modelling can be used to identify candidate release areas and sites based on landscape characteristics, such as habitat type, topography, and human infrastructure, however, this can be complemented by, and can inform, the implementation of ground-based social and ecological surveys, as well as decision-making analysis focused on additional key parameters relevant to the focal species. In addition, expert knowledge (Zamboni, Di Martino and Jiménez-Pérez, 2017; Hunter-Ayad et al., 2020) and data from proxy species (Kalle et al., 2017; Tang et al., 2021) are readily used to fill any areas where knowledge on the species or landscape is incomplete. Assessing suitable reintroduction sites can therefore be a significant investment, both in terms of financial and human resources, and can be time-consuming for highly mobile species, where the scale of assessments is greater (Wakamiya and Roy, 2009; White et al., 2015).

The European wildcat (*Felis silvestris*) is a Critically Endangered species in Great Britain (Mathews and Harrower, 2020), having been described as ‘functionally extinct’ within its last refuges in Scotland because of hybridisation with domestic cats (Breitenmoser et al., 2019). Wildcats have been absent from England and Wales for ~150 years (Langley and Yalden, 1977) because of killing by people, habitat loss and fragmentation (Langley and Yalden, 1977). While many other of Britain's carnivores experience similar pressures, most have recovered to varying extents over the past century, the wildcat however has not (Sainsbury et al., 2019). Alongside hybridisation, roads represent a significant contemporary source of mortality in wildcat populations across their distribution (Bastianelli et al., 2021), while the intensification of agricultural landscapes is also detrimental to their presence (Ruiz-Villar et al., 2023). Assessment of the presence and extent of these factors forms an important part of evaluating if suitable connected release areas can be found when assessing the feasibility of a reintroduction.

Conservation within Scotland is ongoing, with the release of captive-bred wildcats into the Cairngorms National Park (Saving Wildcats., 2023). Additionally, preliminary feasibility assessments for reintroduction to their former native range in England and Wales have also begun (Gow and Cooper., 2018; MacPherson et al., 2020). These initial assessments are informative in furthering the spatial specificity of investigations and highlighting areas of potentially suitable habitat but are conducted at too coarse a scale and are not designed to evaluate release areas or assess potential release sites. Therefore, the next stage in evaluating the feasibility of a wildcat reintroduction is to conduct a fine-scale spatial analysis to evaluate connectivity within and between suitable habitat patches identified by MacPherson et al (2020) to determine the presence of functional release areas. From these outputs, we then seek to identify potential release sites, using an Analytical Hierarchy Process (AHP) decision-making analysis (Saaty 1980) to evaluate and rank each site based on their functionality and the presence of potential threats.

Methods

Study regions

Study regions were determined by the recommendations of MacPherson et al (2020). MacPherson et al used maximum entropy (MaxEnt) niche-based modelling based on data from genetically verified wildcat presence locations from surveys across France to develop a landscape scale model to identify regions of potentially suitable habitat for wildcats in England and Wales at a resolution of 10 km^2. The study proposed three regions for further investigation

– North Wales, West Wales and South West England. To define the spatial limits of these study regions, we used the MaxEnt outputs for each proposed region, and placed an additional 10 km^2 buffer around each output. This increased the chances of including smaller patches of potentially suitable habitat from proximate areas and to better explore connectivity within and between the areas identified by MacPherson et al (2020) at a fine scale. The final three study regions comprised 3575 km^2 in North Wales, 11,463 km^2 across Mid, South and West Wales (hereafter West Wales) and 12,325 km^2 in South West England (Figure 5.1). The landscape in the study regions comprised a mixture of semi-natural habitats and modified land-cover types, including forestry and agricultural land, with small pockets of densely populated settlements and high-traffic roads intersecting what are otherwise rural landscapes.

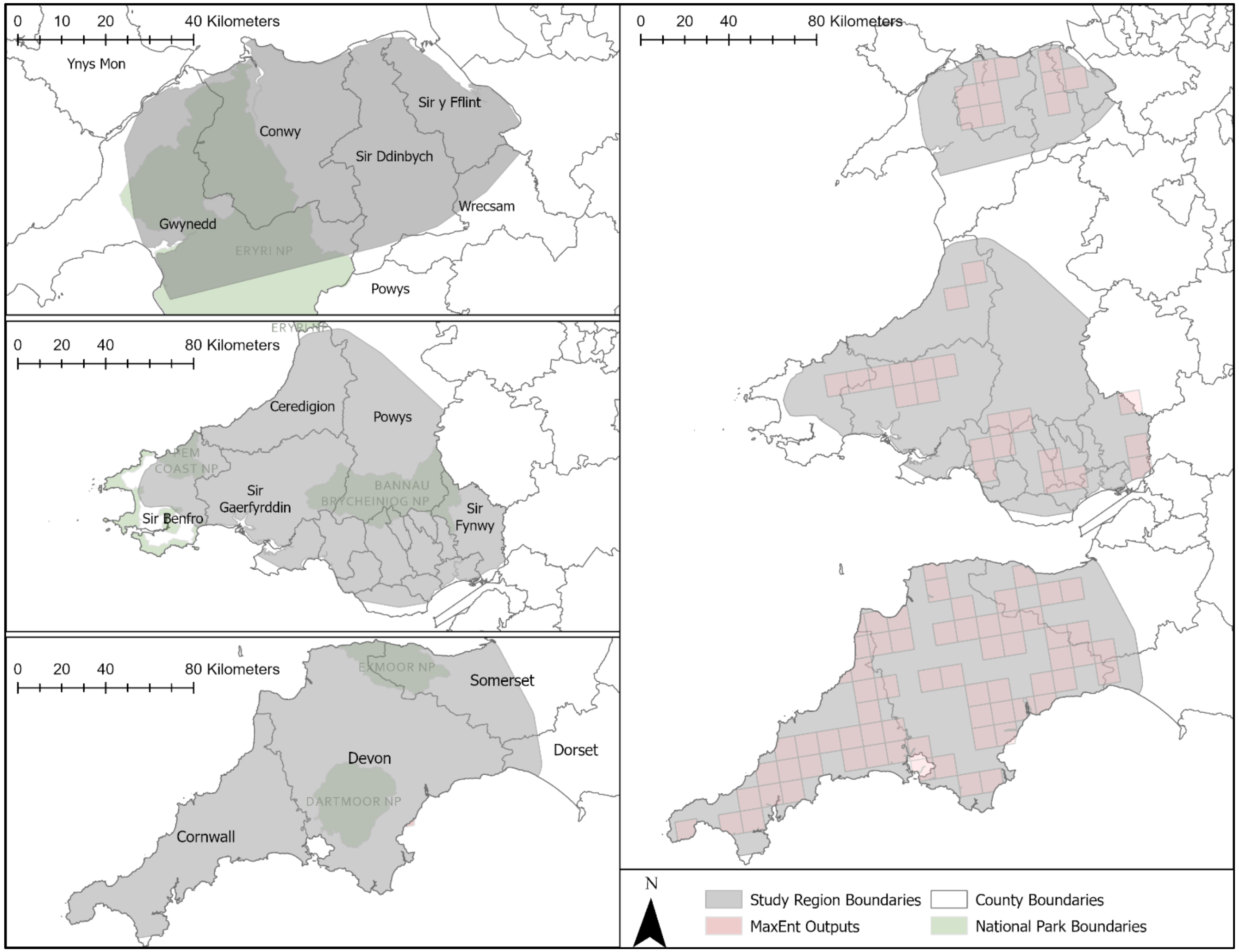

Figure 5.1 Map displaying the extent of study regions (grey) and county and national park boundaries in North Wales (Top left), West Wales (middle left), and South West England (bottom left) and their location within the UK in relation to the MaxEnt outputs from MacPherson et al (2020) (right)

Modelling landscape connectivity

To map connectivity across our study regions, we used Linkage Mapper (McRae and Kavanagh 2011) in ArcGIS Pro. Linkage mapper integrates least-cost path (LCP) analysis with circuit theory, using a landscape resistance surface to map corridors and LCPs between pairs of selected adjacent nodes (cores), creating a network of cores using adjacency and distance data. From this analysis cost-weighted (CWD) and Euclidean (EucD) distances as well as least-cost paths (LCP) were calculated. The resulting outputs are then combined into a single composite map detailing least-cost corridors across a landscape. Additionally, clusters of neighbouring cores based on a maximum CWD, were mapped to form 'constellations' of cores within the landscape. For each study region we determined the spatial extent of the largest cluster of connected cores to be the most suitable candidate release area.

To identify the potential wildcat carrying capacity of resulting connected clusters we calculated the mean home range size with 95% confidence intervals of an adult female wildcat based on radio and GPS tracking data from studies across Europe. This resulted in a mean home range of 5.58 km^2 ±2.284, equivalent to a density of 1.79 wildcats per 10 km^2. This density is lower than most estimates from Germany (Götz et al., 2008), Switzerland (Nussberger et al., 2023) France (Beugin et al., 2016), Italy (Anile et al., 2014; Fonda et al., 2021), Slovenia (Nogueira 2021) and north east Scotland (Kilshaw et al., 2015), but higher than those from the Iberian peninsula (Gil-Sánchez et al., 2020; Matias et al., 2021), Poland (Okarma et al., 2002) and western Scotland (Scott et al., 1993).

Using the 2020 UK Centre for Ecology and Hydrology landcover map, we identified areas of broadleaf woodland <1 km^2 to act as cores. Moreover, where coniferous and broadleaf woodland patches were connected, their combined extent was <1 km^2 and at least 20% of the combined patch extent was broadleaf woodland, the patch was also taken forward as a core (hereafter woodland core area (WCA). The presence of broadleaf and mixed woodland is a predictor of wildcat habitat use, and small woodland patches (>1km^2) can be an important habitat for wildcats, particularly in connecting open areas (Jersoch et al., 2017). In total, 228 WCAs were identified in West Wales, 40 in North Wales and 175 in South West England (Figure 5.2). Linkages between WCAs were not mapped when at a distance of ≥10 km EucD, as there is little evidence of wildcat crossing unforested areas of more than this distance (Klar et al., 2012). Additionally, WCAs were determined to be part of the same cluster when linkages between them were less than or equal to a CWD of 30 km.

Following corridor mapping we calculated current flow centrality, using the Centrality Mapper tool within the Linkage Mapper package. This uses Circuitscape (McRae et al. 2008) to estimate the importance of mapped linkages and WCAs to overall connectivity, by modelling movement as a function of current flow centrality through the LCP network, with the amount of flow dependent on the resistance of individual cells within it. Finally, we utilised the Pinchpoint function in Linkage Mapper which uses Circuitscape (McRae et al. 2008) to identify potential bottlenecks within the corridor networks where movement is likely to be restricted or alternative linkage pathways are unavailable. These areas are important to identify to direct management efforts to areas where connectivity could be compromised

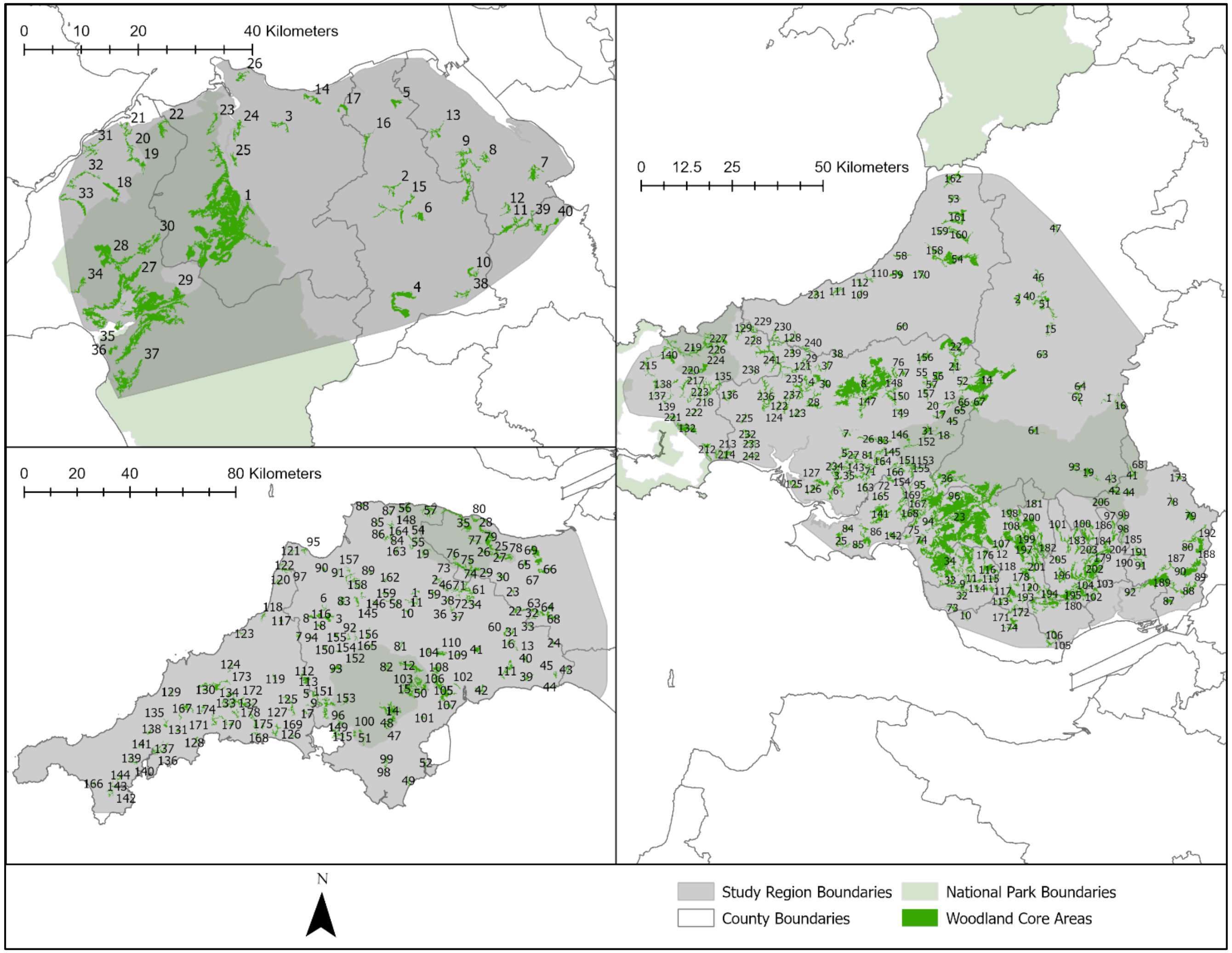

Figure 5.2 Map displaying the distribution of (numbered) woodland core areas (WCAs) in North Wales (Top left), West Wales (Right) and South West England (Bottom left) derived from Centre for Ecology and Hydrology 2020 landcover data.

Landscape resistance map

To create resistance maps for the European wildcat we made use of existing literature and habitat models to derive important landcover characteristics and cost values. The base of our resistance map used the 21 landcover classifications described by the 2020 UK Centre for Ecology and Hydrology landcover map (Morton et al., 2021). These were grouped into 10 classes according to the similarity of their effects on wildcat space use. These classes were broadleaf and mixed woodland, coniferous woodland, arable, grassland, riparian and wetland, heather and heather grassland, coastal and inland rock, small settlements, large settlements, and saltwater (Table 5.1).

In addition to these 10 classes, we defined a further six classes related to road networks (Table 5.1). European wildcats have been shown to avoid primary and multi-carriageway roads at 200 m (Klar et al., 2009), moreover, high-traffic roads (<10,000 vehicles per day) are a significant cause of wildcat mortality and hindrance to dispersal (Bastianelli et al., 2021; Westkemper et al., 2021). Consequently, we placed a 200 m buffer around all motorways (main road for fast-moving traffic with multiple lanes), dual carriageways (multi-lane A-roads intended to provide large-scale transport links), and high-traffic A-roads (single-lane major roads intended to provide large-scale transport links). We reduced the extent of this buffer to 100 m and 50 m for medium (5,000-10,000 vehicles per day) and low traffic (>5,000 vehicles per day) A-roads, which are thought to have a reduced impact. B-roads (smaller connecting roads which feed traffic between A roads and minor roads) were included in the final resistance raster without a buffer.

A further two classes were described based on the impact of settlements on wildcat movement (Table 5.1). Klar et al (2008) found that wildcats avoid larger settlements at 900 m and smaller settlements at 200 m, consequently, we created two classes by creating buffers to reflect these values (Major settlements edge and Minor settlements edge). Finally, while broadleaf and mixed woodland are the primary habitats used by wildcats, open habitat at the forest edge is often utilised for hunting (Klar et al., 2008; Kilshaw et al., 2016). We created the final two classes to reflect this, placing a 200 m internal and external buffer around the edge of broadleaf (Broadleaf edge) and coniferous woodland (Coniferous edge) patches. The final resistance raster was mapped at a 25 x 25 m resolution, where each cell was classified based on which habitat class covered the highest proportion of the cell.

The cost value of each habitat class was determined by an examination of wildcat literature on both habitat use and threats. In the final cost raster, the class with the lowest cost to wildcat movement broadleaf and mixed woodland was given a value of 1 (Table 5.1). Similarly, the importance of edge habitat was reflected by broadleaf and coniferous edges having the next lowest cost values. In contrast, major settlements are impermeable to wildcat movement and, therefore, were given a maximum cost value of 1000, while minor settlements and motorways were given a cost value of 500, as while they represent both a significant barrier and risk, wildcats are able, albeit rarely, to cross them (Klar et al., 2009).

Model sensitivity

To account for spatial variability in our assumptions on habitat quality within classifications, we ran four models which varied the cost values of open and agricultural land, and road networks. This allowed us to investigate the sensitivity of each study region to these changes which are likely to be significant in what are largely human-dominated landscapes. The ability of wildcats to exploit open habitats is determined largely by prey availability (Matias et al., 2021), the structural complexity of vegetation, for example, hedges, copses and orchards (Jerosch et al., 2017; Jerosch et al., 2018) and agricultural intensity (Ruiz-Villar et al., 2023). Our model cannot discern fine scale changes in these structures which may indicate the intensity of activities or the presence of features which increase prey abundance and movement opportunities. Moreover, while the relationship between road density and mortality is an increasing area of research. Bastianelli et al (2021) showed that an increase in the density of high-traffic roads in a home range by 1 km/km^2 can increase the risk of mortality ninefold. The structural characteristics of roads and their role as a barrier to movement are less well defined. Therefore, testing the sensitivity of each study region to changes in the cost of traversing these habitats allows us to provide different scenarios to be used by practitioners interested in planning a wildcat reintroduction.

Table 5.1 Summary table of land-cover classes showing the per cell (25x25m) travel costs assigned to each land-cover class used to create the resistance map for each scenario. (a) open and agricultural classes = low cost, road classes = low cost; (b) open and agricultural classes = high cost, road classes = low cost; (c) open and agricultural classes = low cost, road classes = high cost; and (d) open and agricultural classes = high cost, road classes = high cost.

	Scenario A	Scenario B	Scenario C	Scenario D
Broadleaf and mixed woodland	1	1	1	1
Broadland and mixed woodland edge	2	2	2	2
Coniferous woodland edge	3	3	3	3
Arable	5	15	5	15
Coniferous woodland	10	10	10	10
Grassland	10	20	10	20
Riparian and wetland	15	15	15	15
Coastal rocks and sediment	25	25	25	25
Heather and heather grassland	25	25	25	25
B roads	25	25	50	50
Low traffic A-roads +50 m	50	50	100	100
Medium traffic A-roads +100 m	100	100	200	200
High traffic A-roads + 200 m	200	200	400	400
Minor settlements edge	250	250	250	250
Dual Carriageways + 200 m	350	350	700	700
Minor settlements	500	500	500	500
Motorways + 200 m	500	500	1000	1000
Major settlements edge	500	500	500	500
Major settlements	1000	1000	1000	1000
Saltwater	1000	1000	1000	1000

Analytical Hierarchy Process

WCAs were ranked based on their suitability as a reintroduction site by an Analytical Hierarchy Process (AHP) (Saaty 1980) using SpiceLogic software. AHP is a structured technique for organising and analysing complex decisions. First, this process requires the selection of criteria relevant to the decision-making process. Six criteria were selected (Table 5.2). WCA size (km^2) and WCA connectivity (centrality) reflected the IUCN guidelines for release site selection, suggesting that the size of a release site and the dispersal of animals from it are paramount in selection (IUCN., 2013). The next four criteria were the proportion of optimal habitat (broadleaf and mixed woodland, and woodland edge), as well as three variables relevant to primary threats, domestic cat density, game bird and poultry density, and road density. For these criteria, we included data from within the WCA and a 5 km^2 surrounding buffer to assess the suitability of the surrounding landscape in the decision-making process. Once criteria were chosen, the AHP uses a pairwise comparison matrix to rank each variable against each other in turn based on their influence on the selection of a release site. These decisions were based on existing literature on the relative importance of each threat criterion and the IUCN guidelines for site selection. From this, a relative weighting is calculated for each criterion.

For the second stage of the AHP, alternatives (release sites) are selected to be assessed based on the first stage criteria. For both the 1st stage and 2nd stage process we selected seven or fewer criteria and alternatives as recommended by Saaty (2003) to reduce inconsistency. The selection of release sites for consideration was first, based on their distance to large settlements, motorways, dual carriageways, and high traffic volume A-roads. Known avoidance distances to these features were doubled to remove lower quality release sites. Any WCA within 1.8 km of a large settlement and 400m of a motorway, dual carriageways or high traffic volume A-road were removed. The remaining WCAs were then ranked by multiplying their size and centrality value, ensuring that the largest and most important WCA for connectivity were prioritised. The top seven ranked WCAs were taken forward to be assessed using the 2nd stage AHP framework. Using a pairwise comparison matrix, chosen sites were then ranked against each other for each 1st stage variable in turn. The consistency of the pairwise comparisons was then checked through the calculation of a consistency ratio that must fall below a threshold of 0.1.

Table 5.2 First stage criteria used for the AHP analysis and their sources

AHP Criteria	Measurement	Data source
Size	Woodland core area size (km^2)	CEH 2020 Land Cover Map (Morton et al., 2021)
Connectivity	Woodland core area centrality value	Centrality mapper analysis outputs
Proportion of optimal habitat	Percentage of broadleaf and broadleaf and coniferous edge within a 5 km^2 buffer around release site	CEH 2020 Land Cover Map (Morton et al., 2021)
Game and Poultry density	Holding densities of game and poultry within a 5 km^2 buffer around release site	APHA livestock demographic data: Poultry population report 2022 (APHA 2022)
Road density	Densities of motorways, dual carriageways, and A-roads (km/km^2) within a 5 km^2 buffer around release site	OS-Open roads 2022 (Ordnance Survey 2022)
Domestic cat density	Cat population density ($cats/km^2$) within a 5 km^2 buffer around release site	APHA Cats per squared kilometre dataset (Aegerter et al., 2017)

Results

Largest cluster extent

The initial analyses of the three study regions showed model WW-A had the largest connected cluster of WCAs across all modelled scenarios (Table 5.3). When looking at model averages West Wales also had the largest cluster extent. This was followed by South West England, and North Wales which had the smallest connected clusters (Table 5.3).

Sensitivity to the permeability of agricultural habitats and road networks varied between regions (Table 5.3). In West Wales, greater sensitivity to road networks was found. For models WW-A and WW-B where permeability through roads was higher, the largest cluster sizes were 41% and 37% larger than for WW-C and WW-D where the permeability of road networks was reduced. Moreover, the number of connected clusters found in the study region changed from 5 (WW-A) and 7 (WW-B) to 16 (WW-C) and 18 (WW-D), showing an increase in fragmentation when roads represented a greater barrier. Sensitivity to changes in the cost of agricultural habitats was also shown but these were less substantial than sensitivity to road networks. The extent of the largest cluster declined between the high and low-cost agricultural models by 14% between WW-A and WW-B, and 9% between WW-C and WW-D. Cluster fragmentation primarily occurred in the south east of the study region in areas with higher densities of primary roads and settlements. Specifically, fragmentation along a particular main road (the A4065 from Neath to Merthyr Tydfil,) in WW-C and WW-D accounts for the significant change in cluster extent between models (Figure 5.3).

In North Wales, greater sensitivity to the permeability of open and agricultural land was displayed (Table 5.3). For models NW-A and NW-C where permeability across agricultural land was increased, the largest cluster sizes were 63% and 67% greater when compared to NW-B where the permeability of agricultural land was reduced. Despite the decline in overall extent, the number of connected clusters found in the study region showed only small changes. Sensitivity to changes in the cost of road networks was also shown but these were less substantial. The extent of the largest cluster declined by 2% (NW-A and NW-C) and 13% (NW-B and NW-D) between the low and high-cost road network models. The western edge of the study region, within Eryri National Park, is present in the largest cluster for all models, with fragmentation occurring between this area and central clusters between modelled scenarios, while the east of the study region is isolated throughout (Figure 5.4).

Table 5.3 Model outputs detailing the number of connected clusters and the largest connected cluster (i.e candidate release area) for each study region and modelled scenario - (xx-A) open and agricultural classes = low cost, road classes = low cost; (xx-B) open and agricultural classes = high cost, road classes = low cost; (xx-C) open and agricultural classes = low cost, road classes = high cost; and (xx-D) open and agricultural classes = high cost, road classes = high cost. The table highlights the estimated wildcat carrying capacity (K) and domestic cat density within the identified largest connected cluster to assist discussion around establishing a self-sustaining viable population and hybridisation risk.

Study Region	Model	Number of clusters	Largest cluster size km^2	Wildcat K of the largest cluster	Domestic cat density/km^2 within the largest cluster
West Wales	WW-A	5	5994	1074 (762-1818)	21.6
	WW-B	7	5134	920 (653-1557)	23.2
	WW-C	16	3561	638 (453-1080)	14.8
	WW-D	18	3246	582 (413-985)	16.5
	Model Average	**11.5**	**4484**	**804**	**19.0**
North Wales	NW-A	3	1193	214 (152-362)	26.1
	NW-B	5	437	78 (56-133)	24.4
	NW-C	3	1171	210 (149-355)	25.4
	NW-D	8	379	68 (48-115)	18.5
	Model Average	**4.75**	**795**	**143**	**23.6**
South West England	SWE-A	7	4652	834 (592-1411)	19.3
	SWE-B	10	3166	567 (443-1058)	16.4
	SWE-C	8	4556	816 (579-1382)	19.0
	SWE-D	12	2760	495 (351-837)	17.3
	Model Average	**9.25**	**3864**	**693**	**18.0**

In South West England, greater sensitivity to the permeability of open and agricultural land was displayed (Table 5.3). For models SWE-A and SWE-C where permeability across agricultural land was higher, the largest cluster sizes were 25% and 39% greater compared with for SWE-C, and for SWE-D where the permeability of agricultural land was reduced. Despite the decline in overall extent, the number of connected clusters found in the study region showed only small changes. Sensitivity to changes in the cost of road networks was also shown but these were less substantial than sensitivity to agricultural land. The extent of the largest cluster declined between the high and low-cost road network models by 2% between SWE-A and SWE-C and 21% between SWE-B and SWE-D. For all modelled scenarios the largest cluster is situated within Devon and Somerset, with connectivity into Cornwall to the west and Dorset to the east limited (Figure 5.5). This is primarily due to the density of high-volume roads, including the A30 (between Exeter and Bodmin) which bisects the north and south of the study region and the M5 motorway (between Burham-on-Sea and Exeter) which prevents eastward expansion.

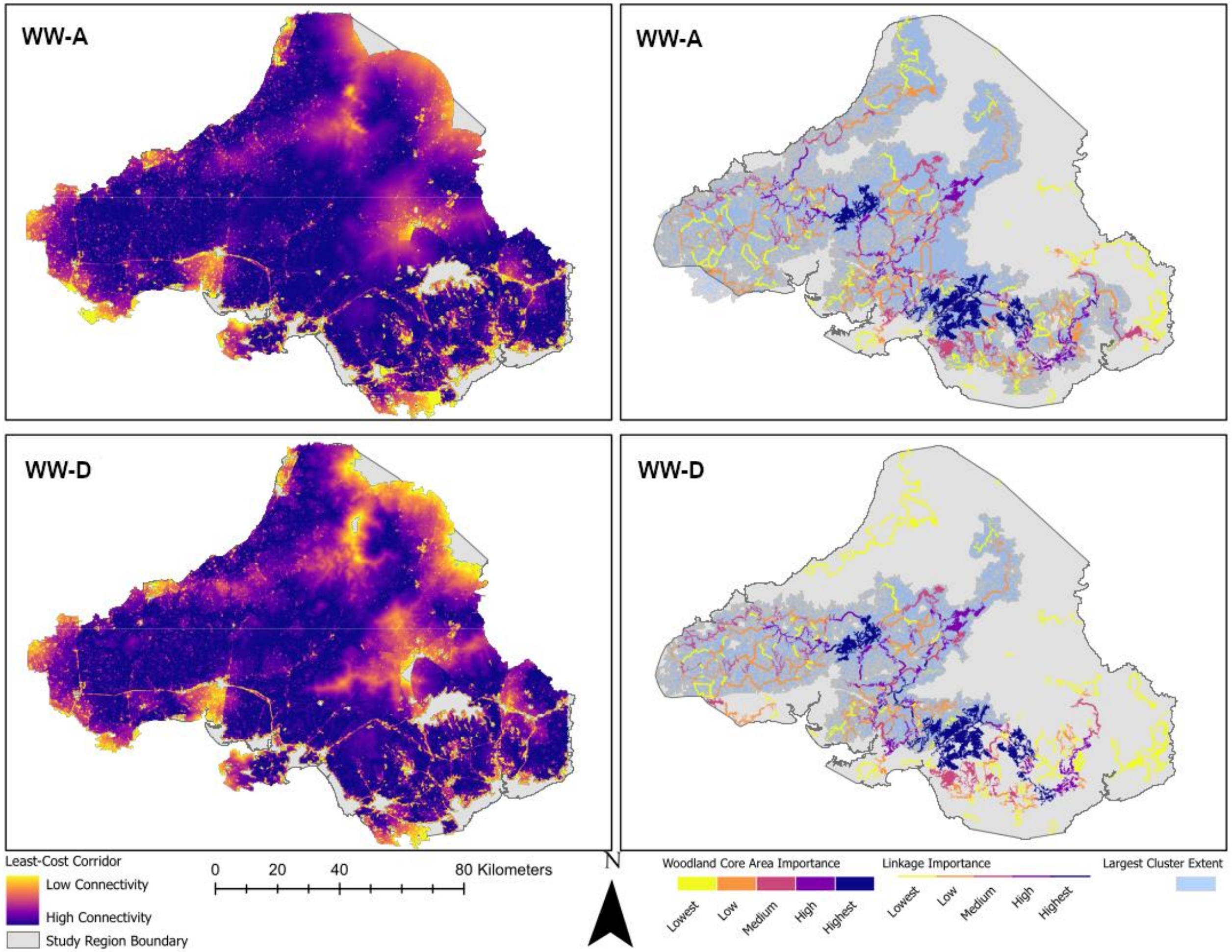

Figure 5.3 (Left) Maps showing the least-cost corridors for West Wales from outputs of the models with the biggest (WW-A) and smallest (WW-D) connected cluster of WCAs. (Right) Centrality scaled WCAs and least-cost path linkages showing the importance of WCAs and linkages for the biggest (WW-A) and smallest (WW-D) connected cluster of WCAs. Corridors, core habitats and linkages are colour-graded according to their centrality score

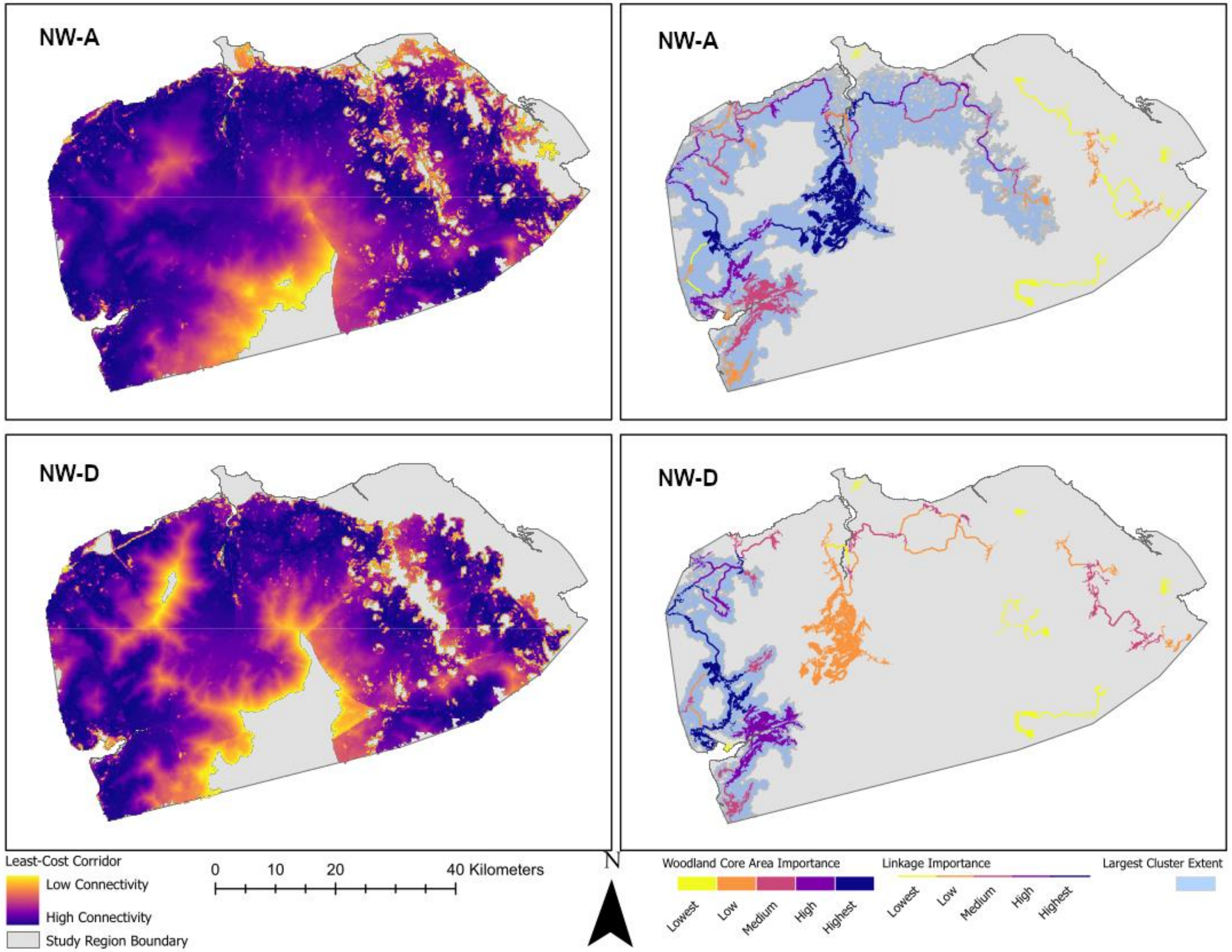

Figure 5.4 (Left) Maps showing the least-cost corridors for West Wales from outputs of the models with the biggest (NW-A) and smallest (NW-D) connected cluster of WCAs. (Right) Centrality scaled WCAs and least-cost path linkages showing the importance of WCAs and linkages for the biggest (NW-A) and smallest (NW-D) connected cluster of WCAs. Corridors, core habitats and linkages are colour-graded according to their centrality score.

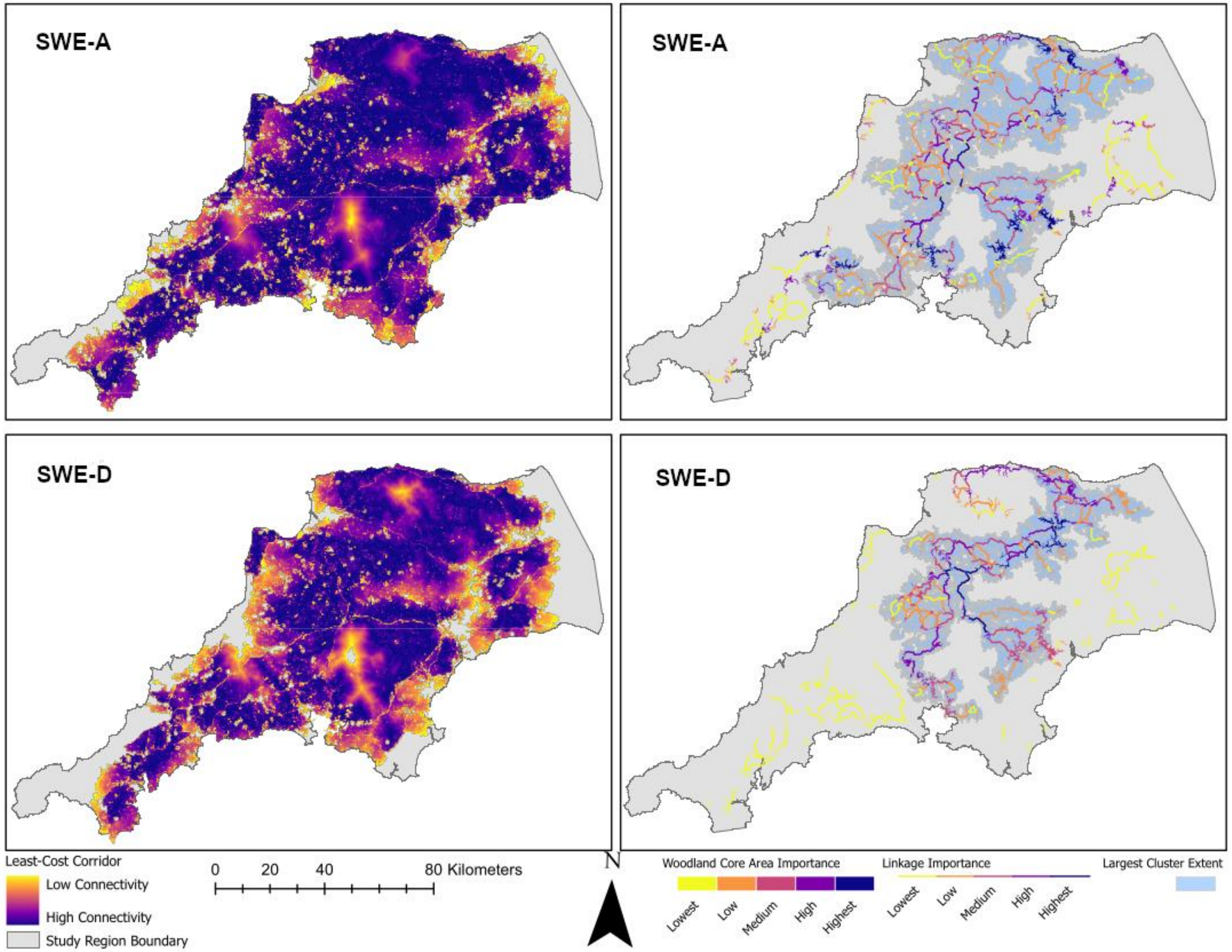

Figure 5.5 (Left) Maps showing the least-cost corridors for West Wales from outputs of the models with the biggest (SWE-A) and smallest (SWE-D) connected cluster of WCAs. (Right) Centrality scaled WCAs and least-cost path linkages showing the importance of WCAs and linkages for the biggest (NW-A) and smallest (NW-D) connected cluster of WCAs. Corridors, core habitats and linkages are colour-graded according to their centrality score

Connectivity between Woodland Core Areas

The connectivity models identified and mapped an average of 371 (354-393) active linkages across the landscape for West Wales, 41 (36-45) in North Wales and 233 (207-259) in South West England between pairs of WCAs (Table 5.4).

Within its largest cluster West Wales had the smallest mean EucD (1.80 km), LCP (2.84 km) and CWD (8.79 km), as well as CWD:EucD (6.20) when compared to South West England and North Wales (Table 5.4). This shows that WCAs are closer together and connected by linkages of higher quality in West Wales. South West England had the largest EucD (3.11 km) and LCP (4.78 km), while North Wales had the largest CWD (15.63 km) and CWD:EucD (7.53), suggesting that while there is a greater distance between WCAs in South West England, the linkages between them are of a higher quality compared to in North Wales (Table 5.4).

Centrality scores varied among WCAs and linkages. In West Wales, both the centrality and area corrected values highlight WCAs distributed between WCA8 and WCA23 as being the most important to connectivity within this study region across all models (Figure 5.3). In North Wales, central and northerly areas of the study region returned higher centrality values for NW-A and NW-C and WCAs along the western edge of the study region scored highest as being important to connectivity for NW-B and NW-D (Figure 5.4). Finally, In South West England, for all modelled scenarios, an area between WCA73 in the north and WCA7 in the south was highlighted as being most important for connectivity (Figure 5.5).

Table 5.4 Model outputs detailing the total number of active linkages mapped between woodland core areas (WCAs) and the number of active linkages within the largest connected cluster for each region and modelled scenario - (xx-A) open and agricultural classes = low cost, road classes = low cost; (xx-B) open and agricultural classes = high cost, road classes = low cost; (xx-C) open and agricultural classes = low cost, road classes = high cost; and (xx-D) open and agricultural classes = high cost, road classes = high cost. The table also provides a summary of the spatial characteristics of active linkages -Euclidean distance (EucD), least-cost path (LCP), cost-weighted distance (CWD), effective resistance and centrality value, to assist comparison of the quality and importance of linkages between models and sites

Study Region	Model	Total Linkages	Linkages in largest cluster	EucD (km)	LCP (km)	CWD	CWD-Euc Ratio	Effective Resistance	Centrality
West Wales	WW-A	393	349	1.79 (0.01-9.84)	2.75 (0.04-18.78)	8.44 (0.05-49.93)	6.31 (1.49-99.75)	460 (1.94-14,608)	1,244 (54-8,240)
	WW-B	364	319	1.67 (0.01-9.84)	2.83 (0.04-18.78)	9.80 (0.05-48.56)	7.62 (1.49-99.75)	583 (2.13-14,620)	1,134 (42-7,558)
	WW-C	374	210	1.96 (0.01-9.80)	2.84 (0.04-18.60)	8.41 (0.04-49.35)	4.87 (1.41-31.74)	197 (3.30-1,503)	1,174 (118-5,857)
	WW-D	354	202	1.76 (0.01-9.80)	2.92 (0.04-18.78)	8.51 (0.05-49.65)	5.98 (1.49-39.60)	387 (2.17-6,162)	1,145 (104-6,648)
	Model Average	**371**	**270**	**1.80**	**2.84**	**8.79**	**6.20**	**406.75**	**1,174**
North Wales	NW-A	45	35	2.89 (0.01-9.01)	4.62 (0.05-14.14)	15.18 (0.07-46.21)	5.98 (2.17-33.04)	424 (1.10-3,266)	424 (12.72-140)
	NW-B	39	19	2.14 (0.01-5.49)	3.39 (0.05-9.59)	14.03 (0.07-31.53)	7.95 (2.17-33.04)	1,121 (1.46-9,958)	24 (7-64)
	NW-C	44	34	2.90 (0.01-9.01)	4.61 (0.05-14.14)	16.62 (0.07-49.83)	7.06 (2.17-58.49)	944 (1.41-14,700)	61 (13-140)
	NW-D	36	14	1.91 (0.01-5.49)	3.59 (0.05-8.34)	16.68 (0.07-48.10)	9.11 (2.02-21.23)	398 (1.31-1,245)	27 (7-56)
	Model Average	**41**	**25.5**	**2.46**	**4.05**	**15.63**	**7.53**	**721.75**	**134**
South West England	SWE-A	259	207	3.38 (0.01-9.85)	4.95 (0.05-19.27)	15.55 (0.10-49.61)	5.48 (1.88-46.60)	579 (3.12-5,955)	597 (12.07-2,970)
	SWE-B	211	154	2.84 (0.01-9.66)	4.43 (0.05-17.56)	15.29 (0.10-45.73)	6.89 (1.91-54.25)	767 (3.52-15,641)	741 (11.50-2,970)
	SWE-C	253	203	3.34 (0.01-9.85)	5.03 (0.05-19.27)	15.53 (0.10-49.96)	5.67 (1.88-46.60)	687 (3.16-11,257)	601 (13.28-2,940)

SWE-D	207	134	2.88 (0.01-9.66)	4.70 (0.05-17.60)	15.55 (0.10-48.67)	6.88 (1.91-72.60)	868 (3.55-15,980)	583 (47-2,475)
Model Average	**233**	**174.5**	**3.11**	**4.78**	**15.48**	**6.23**	**725.25**	**630.5**

The models exhibited the presence of several pinch points in the corridors being mapped. Pairwise pinch points indicate constriction in movement pathways in between the two WCAs which is illustrated by areas with higher current flow. For all models and regions, pinch points were typically located where linkages crossed roads (Figures 5.6, 5.7 and 5.8). Despite this, In West Wales, effective resistance was lowest in models where the cost of traversing roads was greater (Table 5.4). For all regions, few pinch points coincided with important linkages relevant to overall cluster connectivity. Pinch points within the largest cluster in West Wales were predominantly at the southeastern edge of the cluster extent (Figure 5.6). Similarly, the southeast edge of the largest cluster in South West England had the most pinch points, along primary roads (A30 and A377) (Figure 5.8). Within the largest cluster for each region, effective resistance along linkages is lowest in West Wales and highest in South West England (Table 5.4). This indicates that the potential for movement between linkages is greatest in West Wales.

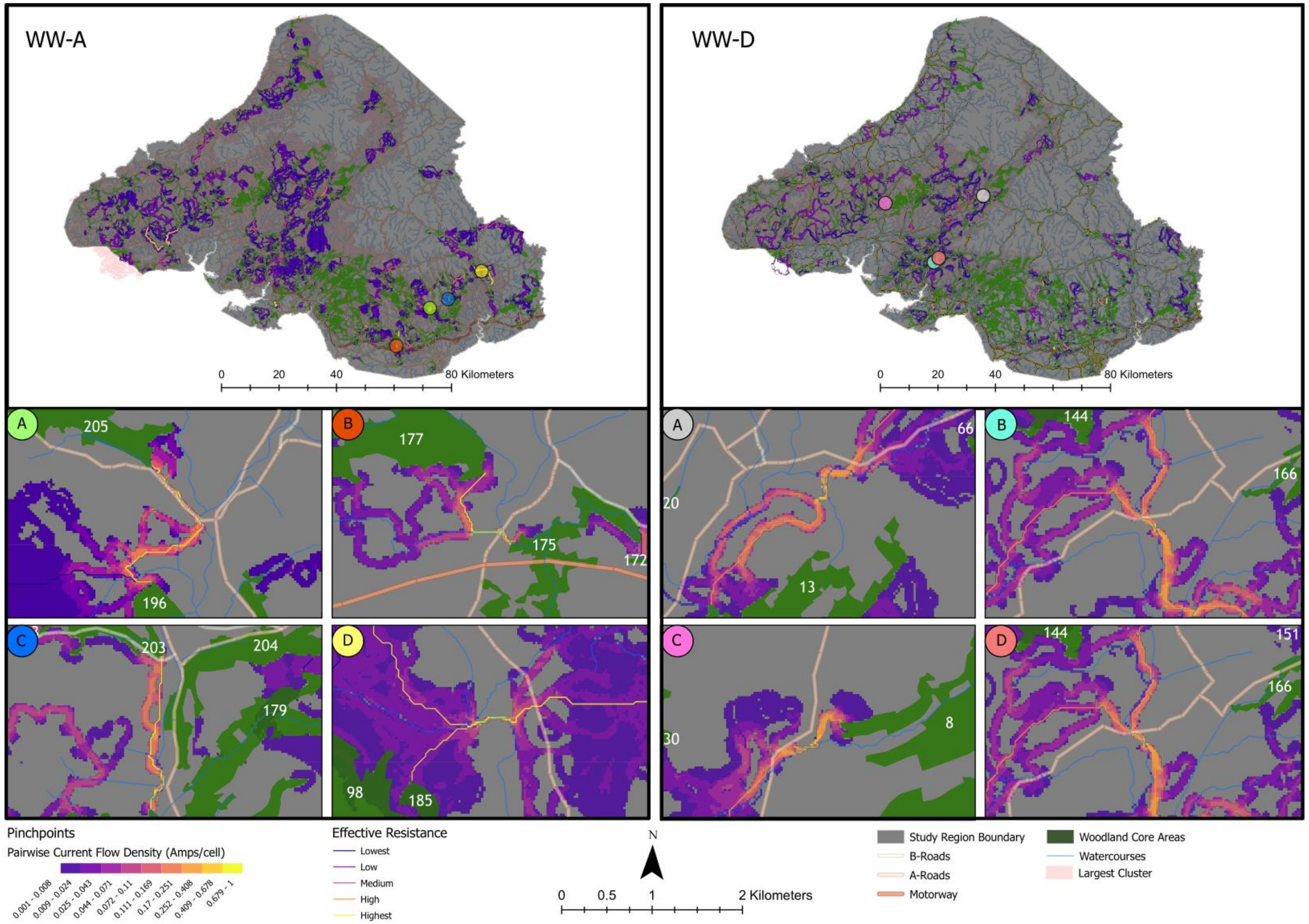

Figure 5.6 (Top) Pairwise pinch points for the models with biggest (WW-A) and smallest (WW-D) connected cluster of WCAs in West Wales; and (bottom) the location of the largest pinch points returned from the pinchpoint analysis corresponding to the coloured circles displayed. Shades of yellow indicate areas where the current flow is highly restricted representing the pinch points

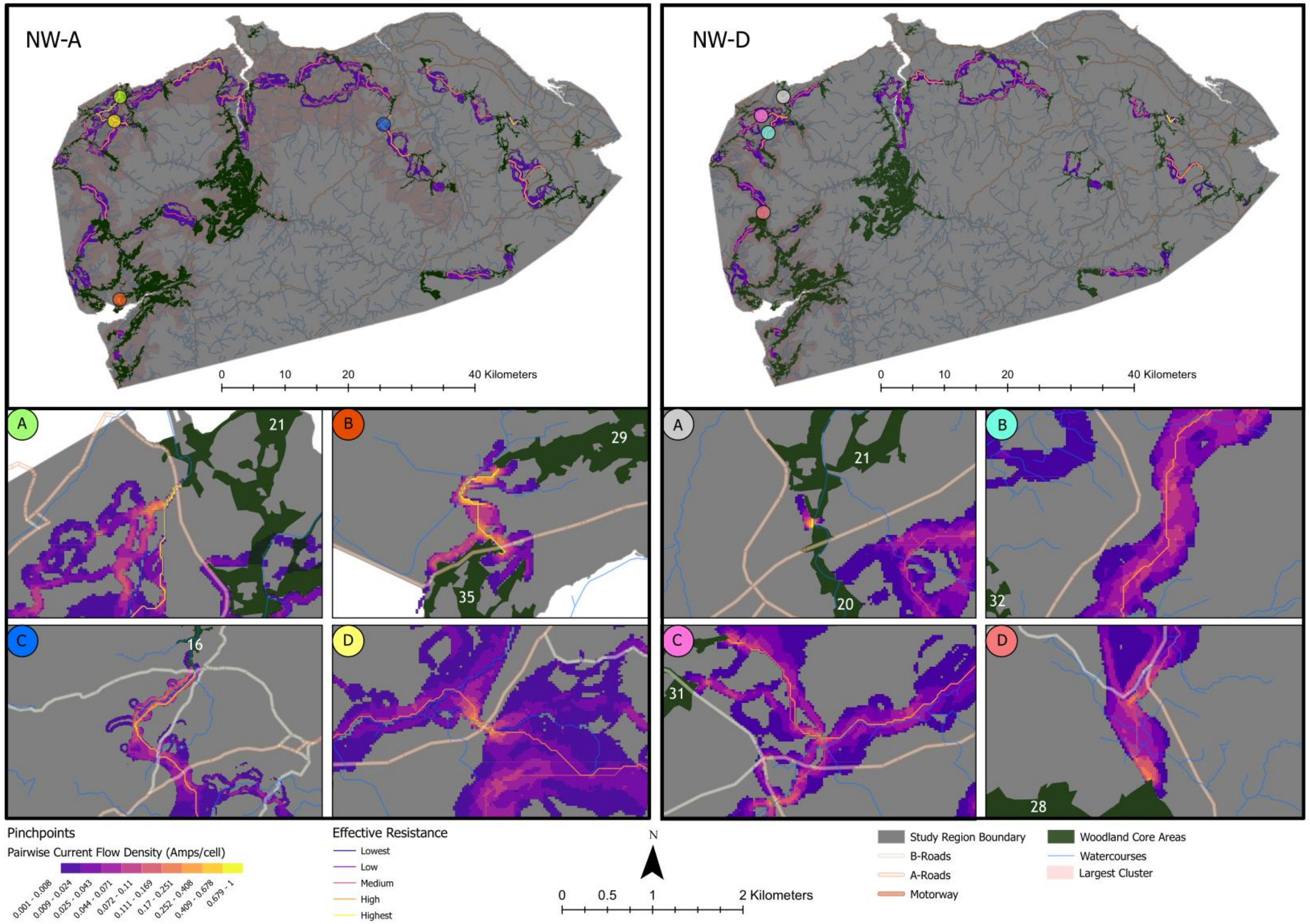

Figure 5.7 (Top) Pairwise pinch points for the models with biggest (NW-A) and smallest (NW-D) connected cluster of WCAs in North Wales; (bottom) the location of the largest pinch points returned from the pinchpoint analysis corresponding to the coloured circles displayed. Shades of yellow indicate areas where the current flow is highly restricted representing the pinch points.

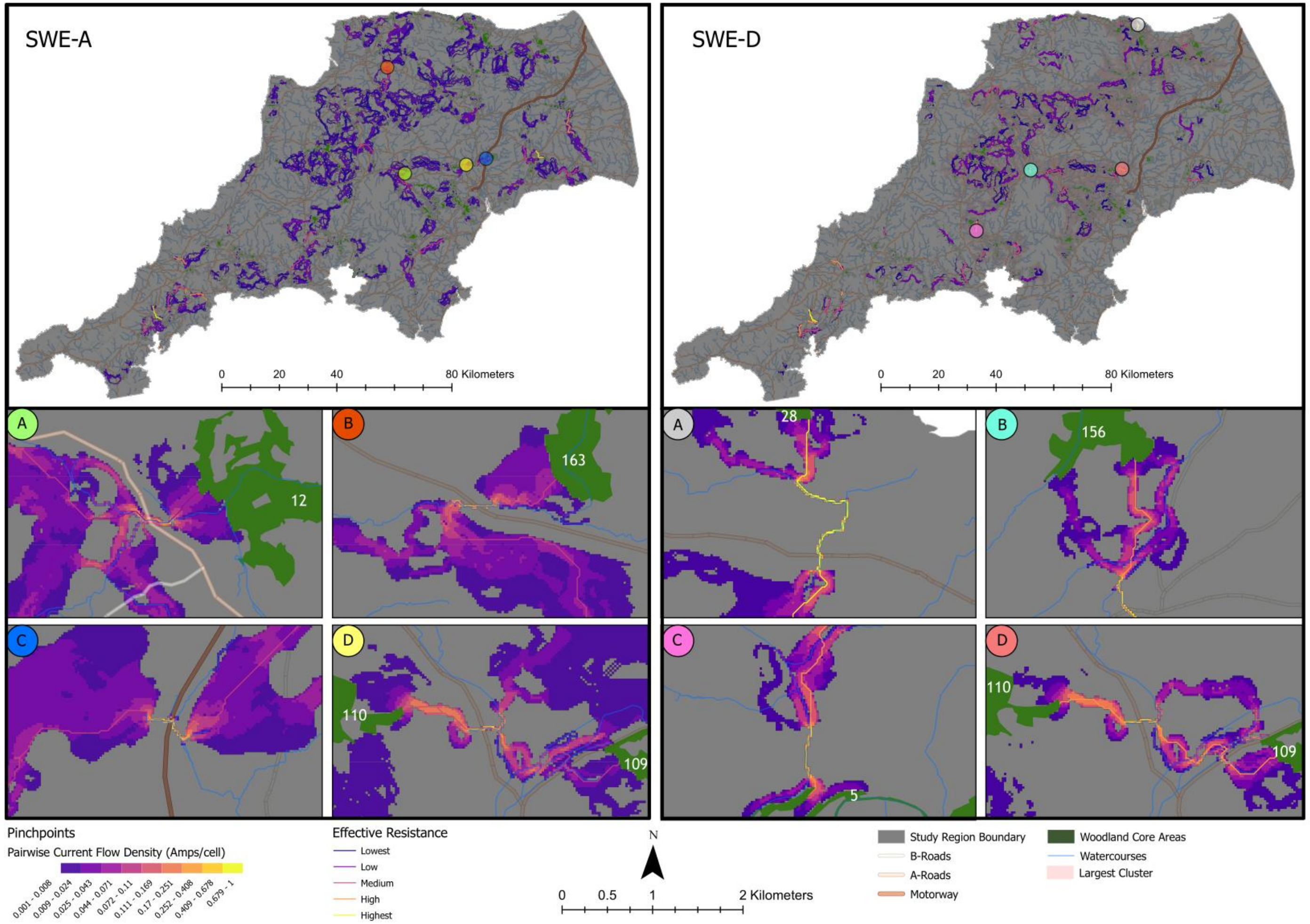

Figure 5.8 (Top) Pairwise pinch points for the models with biggest (SWE-A) and smallest (SWE-D) connected cluster of WCAs in South West England; and (bottom) the location of the largest pinch points returned from the pinchpoint analysis corresponding to the coloured circles displayed. Shades of yellow indicate areas where the current flow is highly restricted representing the pinch points.

AHP

A pairwise comparison of the AHP criteria resulted in domestic cat density and release site connectivity having the equal highest weighting (Table 5.5). This reflected introgression between domestic cats and wildcats being viewed as the primary threat facing a reintroduced population, and the connectivity of the release site being of fundamental importance to the ability of wildcats to disperse into the wider release area as stated by the IUCN guidelines.

From the 228 identified WCAs in West Wales, 99 (43%) met the criteria to be considered as a release site. In North Wales 11 (28%) of the 40 WCAs met the criteria and in South West England 37 (28%) of 130 did. Of the suitable WCAs the top seven based on adjusted centrality were taken forward for analysis via the AHP.

WW8+147 was returned as the most suitable release site for European wildcats in West Wales, owing to its high connectivity and large size, followed by WW148 and WW22 (Table 5.5). NW28 was returned as the most suitable release site in North Wales due to having the highest connectivity value and lowest domestic cat density, followed by NW3 and NW34. Finally, SWE62,73+74 was returned as the most suitable release site for South West England due to its large size and high proportion of optimal habitat, followed by SWE159 and SWE82. When comparing site variables for each criterion, West Wales had the lowest domestic cat density and road density, as well as the highest proportion of optimal habitat and largest potential release sites, while North Wales scored worst for each of those four criteria. North Wales had the lowest density of gamebirds and poultry, and South West England the highest.

Table 5.5 Weighted criteria for each AHP study region and the resulting weighted outcomes for each candidate release site. Bold denotes the highest ranked.

Region	WCA	Connectivity	Domestic cat density	Proportion of suitable habitat	Road density	Game bird and poultry density	Size	Total
	Final criteria weights	**0.305**	**0.305**	0.203	0.102	0.051	0.034	1.000
West Wales	WW8+WW147	**0.126**	0.042	0.039	0.013	0.002	**0.011**	**0.232**
	WW148	0.084	0.053	0.026	0.017	0.008	0.001	0.188
	WW22	0.008	**0.084**	0.013	0.021	0.015	0.006	0.148
	WW4	0.042	0.011	**0.078**	0.004	0.004	0.001	0.139
	WW14+WW67	0.014	0.063	0.016	0.008	**0.019**	0.008	0.129
	WW241	0.021	0.032	0.019	0.013	0.001	0.005	0.091
	WW236	0.010	0.021	0.013	**0.025**	0.001	0.002	0.074
North Wales	NW28	**0.085**	**0.132**	0.056	0.004	0.006	**0.017**	**0.300**
	NW3	0.071	0.008	**0.084**	0.008	0.004	0.002	0.177
	NW34	0.028	0.099	0.005	0.005	0.005	0.001	0.143
	NW37	0.014	0.033	0.028	0.016	**0.019**	0.008	0.119
	NW23	0.057	0.005	0.009	**0.032**	0.003	0.002	0.109
	NW18	0.043	0.016	0.014	0.003	0.005	0.003	0.084
	NW36	0.007	0.011	0.007	**0.032**	0.009	0.001	0.068
South West England	SWE62,73+74	0.077	0.063	**0.023**	0.013	0.006	**0.022**	**0.203**
	SWE159	**0.096**	**0.072**	0.018	0.003	0.001	0.001	0.192
	SWE82	0.039	0.045	0.031	0.026	0.014	0.001	0.155
	SWE113	0.058	0.036	0.013	**0.032**	0.001	0.002	0.142
	SWE12	0.019	0.009	0.092	0.006	0.003	0.005	0.135
	SWE78	0.010	0.054	0.015	0.019	0.009	0.001	0.108
	SWE56	0.006	0.027	0.011	0.002	**0.017**	0.002	0.066

Discussion

The successful establishment of species after their reintroduction depends on the presence of suitable habitat and upon connectivity between suitable habitat patches to facilitate dispersal and gene flow. For the reintroduction of the European wildcat to England and Wales, the presence of dispersal pathways away from potential release sites is additionally important in reducing the risk of interactions between domestic cats and wildcats while a reintroduced wildcat population is established.

The habitat connectivity models, and AHP site selection process identified several key areas that could support wildcat reintroductions in Britain. West Wales emerged as the most suitable region, with the largest connected clusters of woodland, highest quality linkages between woodland patches, and most optimal release sites based on their low domestic cat densities and high site connectivity.

Release area selection

The extent of and connectivity within potential release areas is crucial to the establishment of a released population of wildcats and the long-term success of species reintroduction. West Wales is shown to be the best performing of our three study regions, by having more woodland patches in closer proximity and with lower effective resistance along corridors. Increased connectivity and lower resistance facilitate movement which minimizes risks from inbreeding depression (Wright et al., 2008) as well as increases the opportunities for wildcats to meet conspecifics over domestic cats, therefore reducing introgression risk (Quilodrán et al., 2020; Nieto-Blázquez., et al 2022). Though connectivity was generally poorer in North Wales, the models still identified potential connected areas. However, sensitivity analyses suggest that under scenarios where agricultural land and road densities are a greater cost to movement, the potential carrying capacity is on the cusp of what would be considered viable to establish a genetically diverse population (Littlewood et al., 2014). Therefore, to reduce risks associated with establishing a wildcat population, only West Wales and South-West England should be considered based on this analysis. Connectivity patterns identified in this study can also inform wildcat conservation more broadly. Habitat loss and fragmentation were key contributors to the original extinction of wildcats from the study regions

(Langley and Yalden 1977). The results suggest in the current landscape functional connections exist across two of these regions even amid significant change. To preserve and enhance linkages, conservation actions should focus on restoring habitats and permeability at identified pinch points and between identified clusters. This would expand possibilities for wildcat recovery beyond the largest clusters found here.

Road density

Road mortality is a leading threat to wildcat populations (Bastianelli et al., 2021), and this is likely to be heightened in a reintroduction scenario where released animals are unfamiliar with their environment and risks such as road networks (Kramer-Schadt et al., 2004; Spinola et al., 2008; Robinson et al., 2020; Skorupski et al., 2022). West Wales was shown to have the greatest sensitivity to road networks relative to cluster extent, however, the spatial distribution of road density, means that the loss of extent is concentrated in areas with high densities of settlements in the south-east of the study region. Reduced permeability of some roads may not therefore be entirely negative, as we find that domestic cat densities are substantially smaller within the largest cluster when connectivity with the south-eastern section is reduced by the presence of road networks. Consequently, if roads provide significant barriers to dispersal in these areas, it may reduce possible interactions between wildcats and domestic cats. While road density had less of an impact on the extent of potential release areas in South West England and North Wales, in South West England in particular, pinch point analysis suggests that several connections between the north and south of the release area are bottlenecked to a few narrow road crossings, typically intercepting riverine valleys. Reducing the barriers to road crossing at these points is likely to be important to facilitating the expansion of a wildcat population in South West England. Mitigation strategies such as wildlife crossings and wildcat fencing are effective at reducing the impact of roads on movement and mortality for wildcats (Klar et al., 2009; Klar et al., 2012) and other species (Huijser et al., 2009; Plaschke et al., 2021; Martínez-Medina et al., 2022) and could therefore increase connectivity and the probability of reintroduction success.

Agriculture

Sensitivity to the permeability of open and agricultural habitats is seen most in South West England and North Wales, with both study regions having an increase in the effective resistance and CWD along linkages when the permeability of the agricultural land was increased. However, it is only in North Wales that it is shown to have a significant impact on providing a release area of sufficient size to establish a self-sustaining wildcat population. Wildcats have been shown to thrive in heterogeneous agricultural landscapes which provide food, shelter and cover, and have extensive agricultural practices, (Jerosch et al., 2017). In addition, the presence of linear microstructures (e.g., hedges and tree lines) facilitates the movement of wildcats in such landscapes (Jerosch et al., 2017). Conversely, wildcats have significantly larger home ranges or avoid homogenous landscapes where agricultural intensity is high (Ruiz-Villar et al., 2023). The models used in this study do not have an effective proxy for these microstructures, therefore further investigations into the type of agricultural activities and the structure of agricultural land should be undertaken to refine estimates. The extent of potential release areas in West Wales and South West England was of sufficient size to support a viable wildcat population even when agricultural habitats were more costly to movement, therefore, it may be enough to target future research to the vicinity of potential release sites, rather than the widespread evaluation of the entire release area.

Release sites

The IUCN guidelines (IUCN 2013), state that reintroductions should only be attempted if the original causes of extinction and future threats have been addressed. When assessing release site suitability, candidate sites must therefore balance functionality with threat. The AHP selection process incorporated site connectivity alongside factors like domestic cat densities to identify optimal release sites to begin to assess this. West Wales again ranked highest, with sites scoring better for habitat quality, size and connectivity criteria while minimizing risks such as the presence of domestic cats when compared to South West England and North Wales. Furthermore, West Wales had the largest proportion of WCAs eligible to be considered as a release site, by contrast, South West England, the other region with a potentially suitable release area, had the equal lowest proportion of eligible habitat patches, with most found to be too close to major settlements or primary roads to be considered. These results have a direct impact on the viability of release strategies which can be influenced by the size, number and proximity of release sites and suitable habitat patches (Berger-Tal et al 2011; Helmstedt and Possingham 2016; Stadtmann and Seddon 2018; IUCN 2013).

Wildcats are thought to have low dispersal propensities (Hartmann et al., 2013). In addition, the threat from hybridisation during range expansion (Nussberger et al., 2018; Howard-McCombe et al., 2023) means having a single large release site or multiple proximate release sites is likely to be beneficial for mitigating this risk, by increasing the opportunities to find conspecifics while a core population is established, as well as reducing the chances of problematic hyperdispersal post-release (Berger-Tal et al., 2012) and interactions with human settlements where domestic cat densities are likely to be higher. Potential release sites in West Wales were significantly larger and more isolated relative to human activity and infrastructure than those in South West England and North Wales, allowing greater flexibility in the design of a release strategy. Moreover, candidate release sites in South-West England and North Wales are dispersed more widely across their study region when compared to West Wales, meaning the use of multiple smaller sites may also be limited. Additional research into the short-term establishment and long-term viability of reintroducing wildcats from different sites and over differing periods would be valuable in further refining our findings and helping with devising an appropriate release strategy.

In West Wales, WW8+147 was rated as the most suitable release site owing to its high connectivity and large size. This site along with the second and third-rated sites, WW148 and WW22, provide the large areas of interconnected habitat needed to facilitate a reintroduced wildcat population. In contrast, In North Wales, only NW28 stood out as a potential release site, ranked as the top site due to its high connectivity and low domestic cat density. Overall North Wales sites scored poorly across criteria such as habitat suitability, size, and threats, apart from game and poultry density. This suggests major challenges in establishing a wildcat population compared to the other regions and as with the corridor analysis, point to North Wales being the least likely of the three study regions to facilitate a successful wildcat reintroduction. Finally, in South West England, SWE62,73+74 was rated most suitable due to having a large size and a high proportion of optimal habitat. All sites were ranked closely meaning no single site stood out as exceptional. In addition, the high densities of game birds/poultry exhibited could increase wildcat-human conflicts post-release if this is not effectively mitigated. All favourable sites for each region host either commercial forestry, and/or recreational use resulting in seasonal shifts in traffic volume and disturbance. Local consultation with those who run, work and use these sites is imperative in determining their suitability to ensure that any disturbance and stress to release animals are mitigated and these activities are factored into release planning. Moreover, these results should direct further ground-truthing of identified sites to determine the fine-scale suitability of habitat conditions (Stadtmann and Seddon 2018).

In conclusion, the results indicate West Wales currently presents the most favourable conditions for wildcat reintroduction in Great Britain, outside of Scotland, based on low densities of threats like roads and domestic cats, as well as having the largest and most interconnected habitats. North Wales appears significantly less suitable overall, while South West England offers sufficient habitat extent but higher risks of conflicts and higher cost connectivity. The AHP analysis provides an objective basis for selecting release sites that offer wildcats the greatest chances of thriving. The top sites in West Wales provide a promising starting point for reestablishing wildcat populations. This study provides a quantitative, spatial understanding of habitat networks supportive of wildcat populations to be used at the reintroduction planning stage. Furthermore, the analysed scenarios explore landscape change sensitivities to refine proposals and to be built upon by additional research. Overall, these results suggest West Wales retains the most intact, connected habitat networks, able to sustain recovering wildcat populations.

Chapter Six: Habitat modifies the effects of interspecific interactions between domestic cats and wild mesocarnivores: implications for wildcat conservation and domestic cat management.

Abstract

Domestic cats influence and interact with wild systems. Coexistence between competing mesocarnivores is regulated by niche partitioning of space, time and resources, and so the spatial behaviours of domestic cats are likely to vary depending on interactions with wild species. In this study, we used camera trapping to investigate spatiotemporal interactions between domestic cats and wild mesocarnivores in three landscapes across Great Britain with differing mesocarnivore assemblages. We find co-occurrence between cats and wild mesocarnivores is modified by habitat, but that the influence of covariates differs between sites and species pairs. The presence of badgers and foxes reduced the likelihood of domestic cats occurring away from human habitation in England and Wales, but not in Scotland. Moreover, Wales was the only site where domestic cat occupancy was negatively impacted by increased woodland cover, indicating that site-specific woodland conditions may influence opportunities for domestic cats to exploit woodland environments. We observe significant spatiotemporal overlap between domestic cats and hybrid cats in Scotland, while increased densities of sheep, poultry and game bird holdings are found to reduce co-occurrence between domestic cats, hybrids and wildcats. Our results suggest changes in wild mesocarnivore occupancy and habitat conditions are likely to influence the spatial behaviours of domestic cats. Consequently, changes to habitat, as well as control or restoration of wild mesocarnivores may have wider, unintended consequences by facilitating or inhibiting domestic cat mobility and therefore their ecological impacts, including the likelihood of interactions between cats and wildcats.

Introduction

Domestic cats (*Felis catus)* interact and impact wild environments (Crowley et al., 2020a). Cats are inherently connected to human societies (Slater, 2007; Trouwborst et al., 2020), though they can live along a spectrum of human influence as both companions and feral animals (Crowley et al., 2020a). Cat predation can have potentially deleterious consequences for sensitive species (Loss and Marra, 2017). Moreover, hybridisation between domestic cats and European wildcats (*Felis silvestris)* is an important conservation issue (Howard-McCombe et al., 2021; Tiesmeyer et al., 2020). The extent to which domestic cats impact wildlife and how they are managed are often divisive issues (Gow et al., 2022; Loss et al., 2022; Wald and Peterson, 2020). Both the ecological impacts of domestic cats (Doherty et al., 2017; Loss et al., 2022) and human dimensions of cat management (Crawford et al., 2019; Kennedy et al., 2020; Wolf et al., 2019) have been frequently discussed. Rarely considered are how the ‘wild’ systems that domestic cats inhabit influence these issues, specifically, how interactions with competitive wild mesocarnivores influence the spatiotemporal behaviours of domestic cats and therefore their ecological impact.

Interspecific interactions are essential in discerning spatiotemporal segregation and population dynamics among mesocarnivore communities (Rodríguez et al., 2020; Roemer et al., 2009; Tsunoda et al., 2020). These interactions lead to competitive responses such as kleptoparasitism (Krofel et al., 2022; Prugh and Sivy, 2020), spatial exclusion (Monterroso et al., 2020), intimidation (Ruiz-Villar et al., 2021) and intraguild predation (Palomares and Caro, 1999; Prugh and Sivy, 2020). The outcomes of these behaviours shape the dynamics of communities but also enable them to coexist by facilitating niche partitioning between species and reducing disadvantageous interactions (Prugh and Sivy, 2020; Smith et al., 2018; Tsunoda et al., 2020). Niche partitioning can occur in uses of space (Rodríguez et al., 2020; Schuette et al., 2013), time (Bianchi et al., 2016; Schuette et al., 2013) or resources (Kitchen et al., 1999). Consequently, mesocarnivores often have different habitat preferences and adaptive traits to help minimize risk (Barrull et al., 2014; Bianchi et al., 2016). This can shift between seasons and with increased habitat heterogeneity due to the accessibility of resources (Curveira-Santos et al., 2019; Hernandez-Puentes et al., 2022; Linck et al., 2023). Such separation can occur at a variety of scales, with both landscape and local effects (Barrull et al., 2014; Monterroso et al., 2020).

Anthropogenic activity also plays a significant role in regulating species coexistence and competition. Human presence often results in increased resource availability, creating hot spots where mesocarnivore density, and therefore their interactions, increase (Prugh et al., 2009; Smith et al., 2018). However, increased resource availability can also reduce the competitive nature of these interactions. Furthermore, human infrastructure (e.g. roads, and settlements) (Ruiz-Capillas et al., 2021) and agricultural activity provide both opportunities and barriers to species movement (Gálvez et al., 2021b) and are a primary source of human-induced mortality in mesocarnivores (Beasley et al., 2013; Gálvez et al., 2021a; Prugh et al., 2009). Despite human control and ownership behaviours influencing the spatiotemporal behaviours of many domestic cats (Crowley et al., 2019; Tan et al., 2020), free-roaming cats are also part of a guild of mesocarnivores, and as such would be expected to be exposed to and to some extent regulated by similar ecological drivers of spatial activity as wild mesocarnivores (Prugh et al., 2009). As part of both a domestic and wild system, free-roaming cats also act a conduit for gene flow and disease transmission between the two (Bacon et al., 2023; Howard-McCombe et al., 2021).

The extirpation from Great Britain of larger predators (e.g. Eurasian wolf *Canis lupus* (Yalden, 1999), Eurasian lynx *Lynx lynx* (Hetherington, 2006), Eurasian brown bear *Ursus arctos* (O’Regan, 2018) has altered population and geographic range of some wild mesocarnivore species, shifting their influence on ecosystems (Prugh et al., 2009; Roemer et al., 2009). Conversely, in many places, mesocarnivores are controlled or persecuted, causing widespread depletion of populations and in some instances national extinctions (Curveira-Santos et al., 2019; Reynolds and Tapper, 1996; Sainsbury et al., 2019). The potential for reintroducing lost mesocarnivores and top-order predators has been discussed as a means of restoring populations and ecosystem functions, such as intra-guild competition and top-down trophic cascades (Mills et al., 1993; Wilson et al., 2004)

Studies of cat and wild mesocarnivore interactions have typically focused on urban environments (Carricondo-Sanchez et al., 2019; Louvrier et al., 2022; Mueller et al., 2018; Theimer et al., 2015). In these settings, human pressure causes higher spatiotemporal overlap (Louvrier et al., 2022) while concentrated resource availability can cause an increase in aggressive interactions between mesocarnivores (Theimer et al., 2015). Furthermore, wild mesocarnivores often alter their spatial and temporal use due to disturbance caused by the presence of domestic dogs (*Canis familiaris*) and domestic cats (Carricondo-Sanchez et al., 2019; Gálvez et al., 2021b, 2021a). Red foxes (*Vulpes vuples*) can exploit urban environments more than many mesocarnivores (Mueller et al., 2018; Parsons et al., 2019) and have been shown to have significant spatiotemporal overlap with domestic cats in urban settings (Herrera et al., 2022). The dispersion of human settlements often inflates the densities of domestic and adapted wild species within such systems, limiting the applicability of the results to rural landscapes. In rural spaces, feral cats exploit open habitats and carcasses more readily following fox control programmes (Molsher et al., 2017), while in the presence of foxes, domestic cats avoid areas of pasture (Rodríguez et al., 2020) and areas with greater proportions of natural habitat when fox densities are high (Ferreira et al., 2011). Where present, top-order predators such as wolves, lynx and dingoes (*Canis lupus dingo*) have all been shown to kill domestic cats, however, their effect on cat behaviour is unclear (Allen et al., 2015; Figueiredo et al., 2020; Nájera et al., 2019). Similarly, evidence of top-order predators influencing the spatiotemporal behaviour of wild mesocarnivores is also mixed and likely to be dependent on the availability of prey and quality habitat (Crooks and Soulé, 1999; Ferretti et al., 2023).

In the context of domestic cat-wildcat hybridisation, there is strong spatial segregation between domestic cats and wildcats in some landscapes (Gil-Sánchez et al., 2015; Rodríguez et al., 2020) but greater overlap in others (Beutel et al., 2017). Furthermore, in landscapes where hybridisation rates are high, the overlap of space and resources between domestic cats, hybrid cats (hereafter ‘hybrids’) and wildcats is also high (Biró et al., 2005; Kilshaw et al., 2016). Human settlements largely define where cats are found (Ferreira et al., 2011), with overlap between feral cats and wildcats typically occurring close to farms. Domestic cat tolerance for conspecifics is often determined by resource availability (Turner and Bateson, 2014), meaning cat colonies can form around reliable sources of food and shelter, creating potential ‘hybridisation sinks’. It has been hypothesised that interactions with competitive mesocarnivores and top-order predators are an important factor in segregating domestic cats and wildcats, subsequently reducing hybridization risk (Gil-Sánchez et al., 2015; Rodríguez et al., 2020).

The UK has a national population of 11 million owned cats, while the number of feral animals is unknown (PDSA, 2022). It is also a country of contrast with both depleted and recovering landscapes (Schulte to Bühne et al., 2022; Starnes et al., 2021). Moreover top-order terrestrial predators are extinct from the UK (Yalden, 1999) while mesocarnivore communities are driven by historic predator control and recent reintroductions (Sainsbury et al., 2019). Following the reinforcement of the European pine marten (*Martes martes*) population in Wales (McNicol et al., 2020), a feasibility study for the reintroduction of wildcats to England and Wales is now underway (Breitenmoser et al., 2019; MacPherson et al., 2020). Understanding how interactions between domestic cats and mesocarnivores impact the former's spatiotemporal patterns is fundamental to understanding the spatiotemporal patterns of hybridisation risk for reintroduced wildcats.

In this study, we use single and multi-species occupancy models to explore how anthropogenic and landscape effects as well as interspecific interactions influence cat occupancy and analyse overlap in temporal activity patterns for cats and wild mesocarnivores. We compare landscapes with contrasting mesocarnivore assemblages in England, Wales and Scotland to explore what our results mean for the management of both wild mesocarnivores and domestic cats, species reintroductions, and hybridisation between wildcats and domestic cats.

Methods

Study area(s)

We selected three rural study areas which varied in their mesocarnivore assemblages. The first study area included the counties of Devon (3.577, 51.081) and Cornwall (-4.389, 50.667) in south-west England (hereafter SW England) and was concentrated around Exmoor National Park in an area of 301 km^2; with secondary locations covering an additional 145 km^2. Land cover was a combination of agriculture (~63%) and woodland (~18%) with <1% settlements. The second area comprised 457 km^2 in Carmarthenshire in south-west Wales (hereafter SW Wales), centred on Brechfa forest (-4.031, 52.03). Agriculture (~41%) makes up the highest proportion of the land cover, followed by woodland (~38%), with settlements comprising <1%. The final study area in north-east Scotland (hereafter NE Scotland) was comprised of two sites, the Angus Glens (-3.149, 56.78) and Strathbogie (-2.852, 57.401), comprising 448 km^2 and 568 km^2. The largest proportion of land cover in the Angus Glens was heather and acid grassland (~67%) followed by woodland (~17%), with settlements comprising <1% of total land cover, while Strathbogie comprised agriculture (~44%) and woodland (~31%) with settlements comprising ~1% of total land cover.

Across the sites, five wild carnivoran species, red fox, European badger (*Meles meles*), European pine marten, European wildcat, and hybrid cats (*F. silvestris x F. catus)* were identified, in addition to domestic cats*.* Hybrids were classed as wild animals in this study as they exist outside of human control. At our Scottish site, all five wild species are present. Wildcats and hybrids are absent from the Welsh site, while at the English site, wildcats, hybrids and pine martens are absent. Pine martens were released into Wales between 2015 and 2017, and are thought to have recolonised the study area, while pine marten reintroduction plans for south-west England have also been announced (Devon Wildlife Trust 2022).

Camera surveys

Camera surveys were conducted in SW England and SW Wales between July-October 2021, with additional data from Wales collected between July-August 2020, and in NE Scotland between December-March 2017/2018. The study design was consistent between our English and Welsh study areas but differed at the Scottish sites. In England, 77 cameras were deployed in nine 3x3 clusters (Clark, 2019) and in Wales 99 cameras were across eleven 3x3 clusters. Within clusters, cameras were spaced at intervals of ~600m. This was deemed most appropriate based on the smallest average home range estimates for feral cats in rural environments (1.16 km^2). Spacing cameras at half the home range distance ensures that individuals with the smallest recorded home range had a non-zero probability of encountering a station, with increases in spacing reducing detectability (Clark, 2019; Sollmann, 2018). Predicting detection probability for free-roaming cats is complicated due to the extensive variation in home range sizes that exist (1.16-23.24 km^2) (Bengsen et al., 2016; Hall et al., 2016b). The placement of clusters and camera locations was stratified based on habitat, land use and accessibility of sites. However, due to unforeseen access restrictions, in a few instances, we were unable to set up cameras in every grid cell and at ~600m intervals. Cameras were deployed sequentially, typically four clusters at a time. This approach offered maximum coverage with available resources while maintaining high densities of detectors within clusters to record data at a fine scale. The study design in Scotland differed due to it primarily targeting wildcats. Cameras were deployed using a 1.25 km grid rather than in clusters, with cameras placed at good locations to detect domestic cats if there were suitable habitats within the grid cell and no more than one camera per grid cell. See Campbell et al. (2023c) for details. Data from 117 cameras were used.

In total, 294 cameras (Browning Recon Force Elite HP4, Cuddeback (Black Flash Ambush and Black Flash E3 and Spypoint Force 12)) were deployed across the study areas. Cameras were active for an average of 21 days (SD) and nights in England and Wales and 28 days and nights (SD) in Scotland. Camera traps were attached to suitable trees, fence posts or stakes at 20-40 cm above ground level, to provide the best angle for capturing target species. To keep our focal species in the camera frame for longer and to aid identification, each camera had a stake placed 1.5-3 m in front of it. In England and Wales, each stake was sprayed with a valerian root oil lure and in Scotland dried valerian root was used in conjunction with food bait (gamebirds, usually quail) (Campbell, Langridge, et al., 2023). Each camera station was visited once during the survey period to ensure the camera was functioning as expected and to change batteries and SD cards if needed.

While most target species were morphologically distinct, wild-living cats caught on camera were classified as a wildcat, hybrid or domestic/feral cat based on seven key pelage characters (Kitchener et al., 2005), where a score ≥17 out of a maximum of 21 was defined as a wildcat, <10.5 as domestic cat and the remainder as hybrid (Campbell, Langridge, et al., 2023). In addition, when cats were recorded, attempts were made to distinguish between pet/farm/feral animals by asking households in the vicinity of the camera location.

From each detection, species, date, time, the number of individuals, location, sampling day and sampling occasion were recorded. Data were selected according to a criterion of 30-minute intervals between photographic captures at the same camera trap site of the same species, to ensure data points were independent. Given that most target species are nocturnal or crepuscular each sampling day was defined as starting at noon and ending at 11:59am the following day. This eliminates scenarios whereby an animal visiting a camera trap on either side of midnight would be recorded present during two consecutive sampling occasions. Sampling occasions were set at 1 week in length to reduce the number of observations where the count of species detections was zero while also providing a large enough sample size to effectively model both detection and occupancy.

Occupancy modelling

To analyse the occupancy probability of domestic cats in response to habitat and the presence or absence of the five wild mesocarnivore species we used the R package `Unmarked` (Fiske and Chandler, 2011) to implement single species (MacKenzie et al., 2002) and multi-species (Rota et al., 2016) co-occurrence models.

These models used presence/absence data of each species at each camera station across sampling occasions. The modelling framework uses a multivariate Bernoulli distribution and includes a latent occupancy state (Zi) for each of the species (s) at each station (i), where Zi is represented by a sequence of 0/1 values indicating whether each species is recorded (1) or not recorded (0) at site i. For two species Z is modelled as, $Z \sim$ Categorical ($\psi 00, \psi 10, \psi 01, \psi 11$). Here, the latent occupancy state for all species present is represented as ψ11, when all are absent as ψ00, and only one is present as ψ01, ψ10. The latent states for S species are described by natural parameters (f) which describe the log-odds a species occupies a site. For two species, the natural parameters are f1, f2, and f12. Fixing f12 to zero assumes independence in species occurrence. As in any occupancy model, ψ can be modelled as a function of covariates.

Two levels of modelling were conducted. First, single-species occupancy models were undertaken, with the marginal occupancy of each species at each site modelled as a function of each of the covariates in turn. Secondly, we used multi-species models to analyse conditional occupancy (effects of each species' presence on other species' occupancy) between each wild mesocarnivore and domestic cats. We modelled variation in occupancy probability using the covariates: Building distance; river distance, woodland cover, livestock holding density, elevation, and slope. These are related to factors thought to be relevant to domestic cats and wild mesocarnivores behaviour. All spatial analyses were conducted in ArcMap 10.6.

Temporal activity

To assess the effect of temporal activity on interspecific interactions, we used the R package `Overlap`. Detection times were converted to sun time to better reflect the timing of sunset and sunrise and account for shifts in these throughout the data collection period (Nouvellet et al., 2012). To compare overlap in daily activity patterns, we used a kernel density estimation method for circular data developed using the R package `Overlap` (Ridout and Linkie, 2009; Meredith and Ridout, 2022). Time was converted to radians for each species, before comparing the degree of temporal overlap between species pairs from a probability density curve. Temporal overlap between pairs is estimated with the dhat (Δ) coefficient of overlapping, where a value of 0 represents no overlap and 1 represents complete overlap. There are three variants of the Δ estimator ($\Delta 1$, $\Delta 4$, and $\Delta 5$), it is recommended to use $\Delta 1$ when the number of photographic detections of a species within an area is <50, and $\Delta 4$ when detections are>75 (Meredith and Ridout, 2022). As the number of detections was below 50 for domestic cats in our study areas in England and Wales, the $\Delta 1$ estimator was used for these sites, with $\Delta 4$ used for overlap estimates between species pairs in our study area in Scotland due to detections being >75. We used a smoothing parameter of 1 for $\Delta 4$ and a parameter of 0.8 was applied to overlap estimates using $\Delta 1$ (Meredith and Ridout, 2022).1000 smoothed bootstrap samples were used to generate 95% confidence intervals for each overlap estimate. As the coefficient of overlap is a descriptive statistic, Watson's two-sample U2 test for circular data was performed using the "`circular`" package in R (Lund, Agostinelli and Agostinelli, 2022) to calculate significance estimates between density curves for each species pair.

Results

In total, 1570 independent records of the five wild mesocarnivores and domestic cats were included (England, 313, Wales 494, Scotland 763). There were 28 independent detections of domestic cats at sites in England, 26 in Wales and 78 in Scotland. Detection probability was highest for domestic cats in Wales and lowest in Scotland (Figure 6.1). Foxes had the highest detection probability in both England and Wales, but the lowest in Scotland, where badgers had the highest detection probability.

Single species occupancy models

Mesocarnivore occupancy probability varied between species and sites. Domestic cats had the lowest mean occupancy probability at all sites (Figure 6.1). Mean occupancy probability was highest for foxes in both England and Wales and for pine martens in Scotland. Pine marten had the largest variation in occupancy probability between sites of any species and badger had the lowest variation.

Cat occupancy probability declined with increased building distance in England and Wales (Figure 6.2). In Wales, cat occupancy probability also declined with increased woodland cover. In Scotland, domestic cat occupancy was not influenced by any covariate.

Among wild mesocarnivores (Figure 6.3), fox occupancy probability declined as woodland cover increased in Wales and decreased with increased river distance in Scotland. Badger occupancy probability declined with increased river distance in England and with increased elevation in Scotland, as well as increasing with the density of livestock holdings in Scotland. Pine marten occupancy probability was higher with increased building distance in both Wales and Scotland, as well as with higher woodland cover and elevation in Wales. Wildcat occupancy probability was lower in areas with higher densities of livestock holdings and with increased river distance in Scotland.

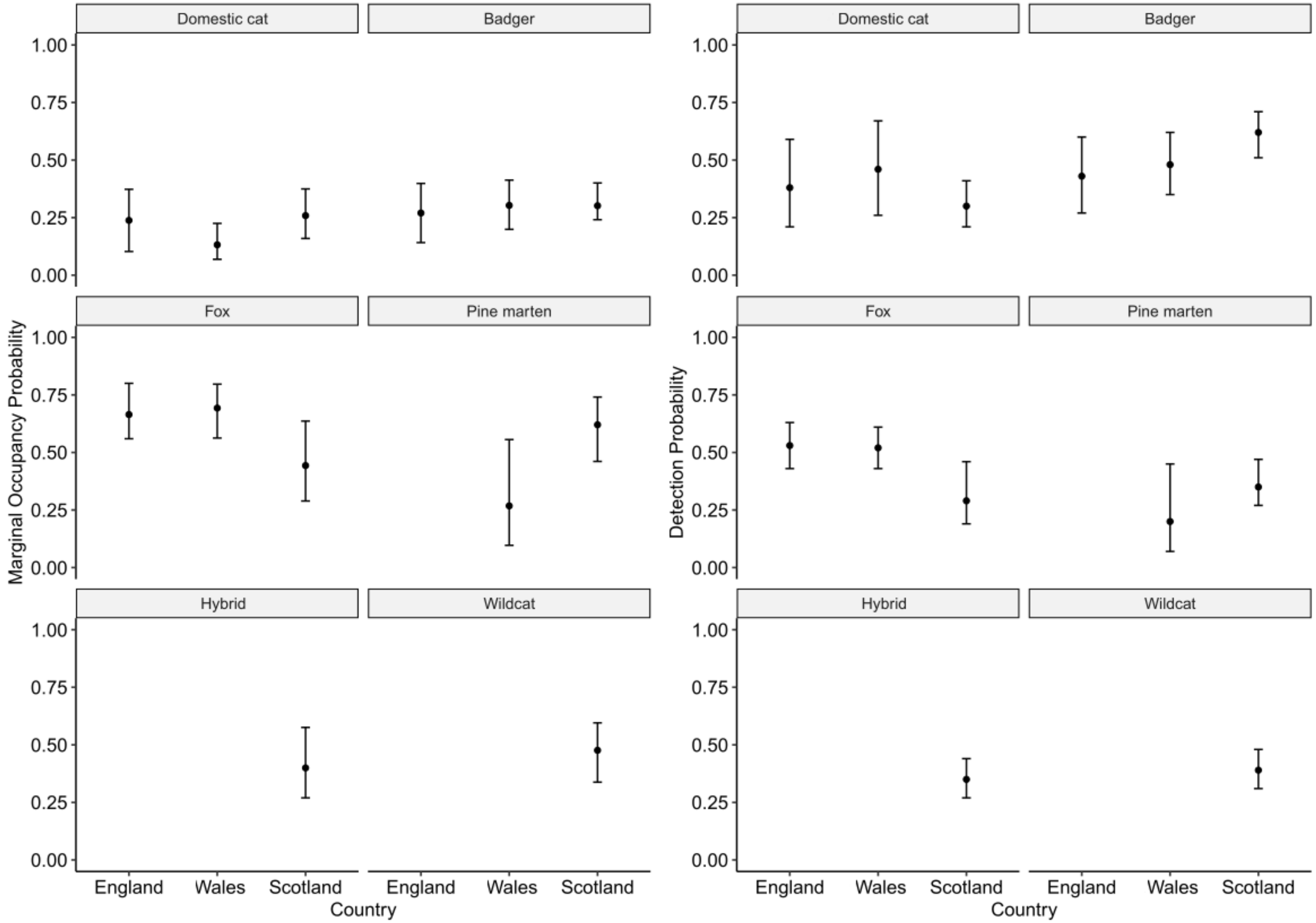

Figure 6.1 Comparison of (left) mean (95% CI) occupancy probability (*ψ*) values for the domestic cat, badger, fox, pine marten, hybrid cats and wildcats in each study area. (right) Mean (95% CI) detection probabilities (*p*) for the domestic cat, badger, fox, pine marten, hybrid cats and wildcats in each study region.

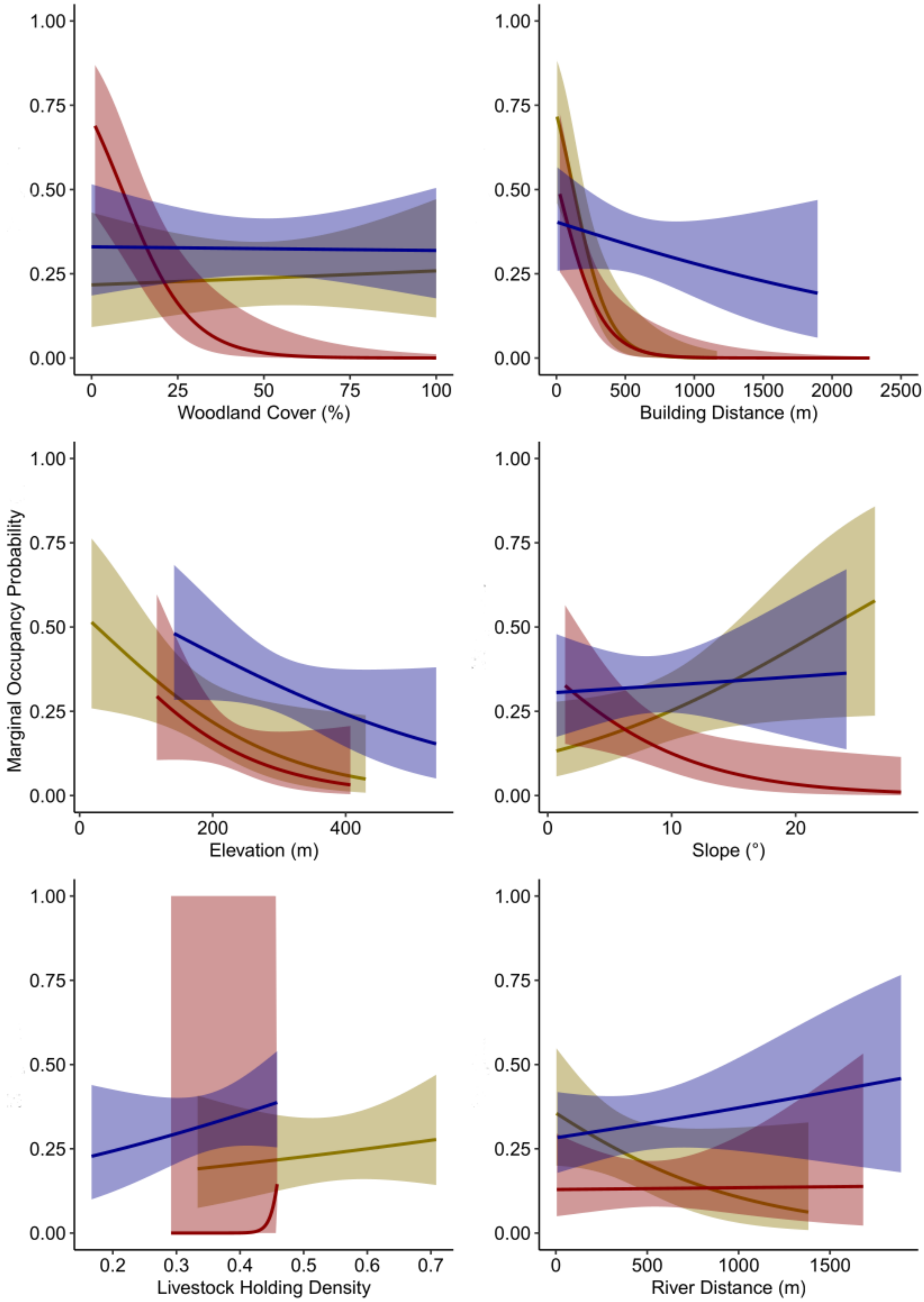

Figure 6.2 Estimated domestic cat probability of site occupancy (ψ) as a function of woodland cover (%), building distance (m), elevation (m), slope (◦), livestock holding density and river distance (m) in all study areas. England = Yellow, Wales = Red, Scotland = Blue

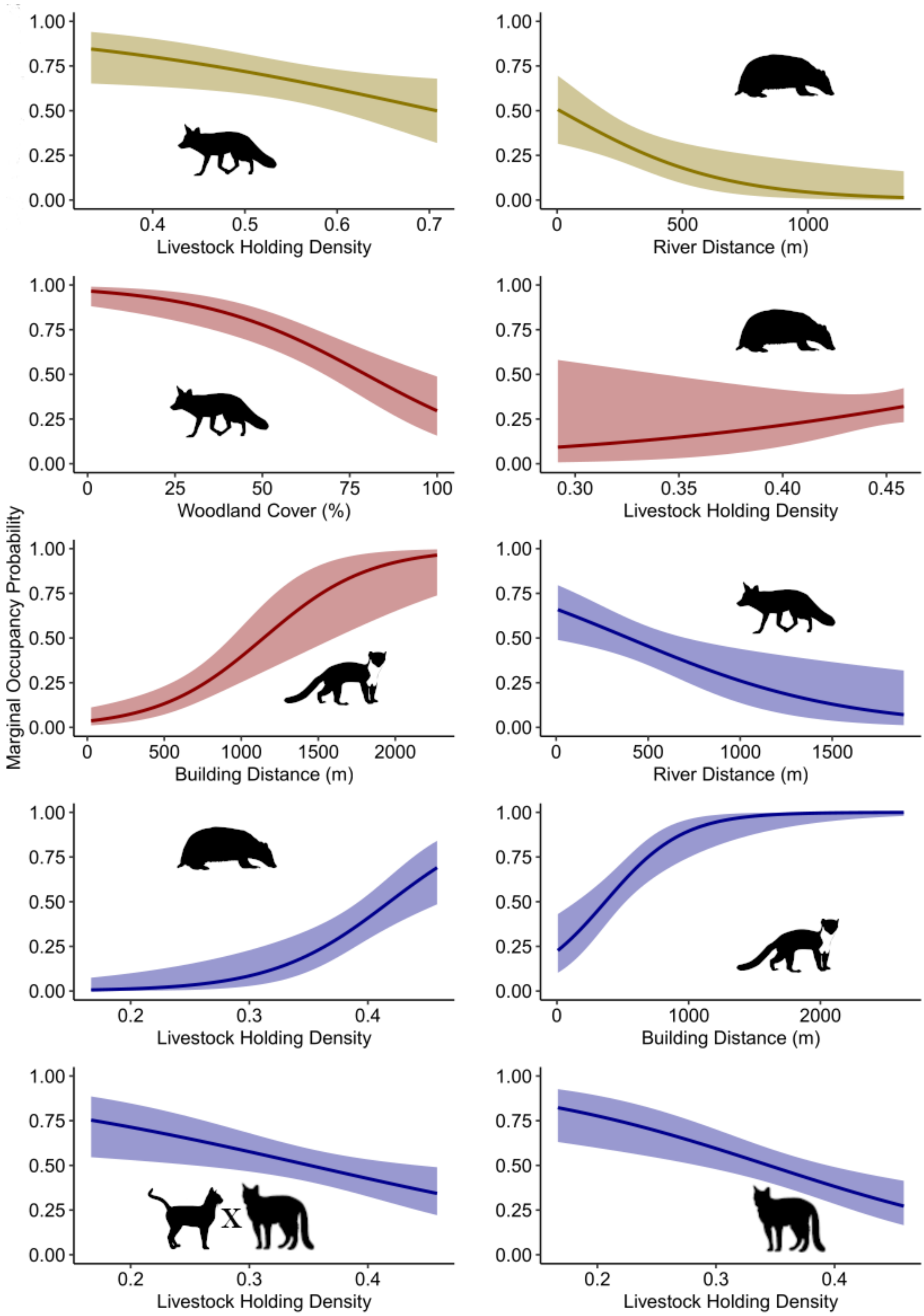

Figure 6.3 Estimated probability of site occupancy (ψ) for fox, badger, pine marten, hybrid cat and wildcat as a function of covariates selected from single species occupancy models with the smallest AIC. Covariates include woodland cover (%), building distance (m), elevation (m), slope (◦), livestock holding density and river distance (m). England = Yellow, Wales = Red, Scotland = Blue

Conditional occupancy

Cats and wild mesocarnivores

When covariates weren't considered, the occupancy probability of domestic cats varied only in the presence and absence of foxes (Figure 6.4a). Conditional occupancy models showed that at our site in England, cats were less likely to occupy a site when foxes were present. When covariates were introduced, we found that the presence of badgers significantly affected the influence of environmental variables on the occupancy probability of domestic cats at all sites: At sites in England and Wales, domestic cats were less likely to occupy sites at increased building distance when badgers were present, relative to equivalent sites where they were absent. Furthermore, in Wales, domestic cats were also less likely to occupy sites with increased woodland cover if badgers were present, compared to when they were absent. In Scotland, domestic cats were less likely to occupy sites at higher elevations and more likely to with increasing livestock density when badgers were present compared to sites where they were absent. In Wales, the presence of foxes affected the influence of environmental variables on the occupancy probability of cats: Domestic cats were less likely to occupy a site with increased building distance when foxes were present relative to equivalent sites where foxes were absent. Finally, in Wales, domestic cats were less likely to occur with increased woodland cover when pine marten was present, relative to sites where they were absent.

Domestic cats, hybrid cats and wildcats

When covariates weren't considered, domestic cats were more likely to occupy a site when hybrids were present (Figure 6.4b). When covariates were introduced, hybrids were less likely to occupy a site with increased livestock holding density when wildcats were present relative to equivalent sites where wildcats were absent. This same relationship is shown for river distance, with hybrids less likely to occur at sites far from rivers when wildcats are present. Finally, when livestock density is low, we also find that domestic cats are more likely to occupy a site when both hybrids and wildcats are present, however, as livestock holding density increases the probability of a domestic cat occupying a site decrease if hybrids and wildcats are present.

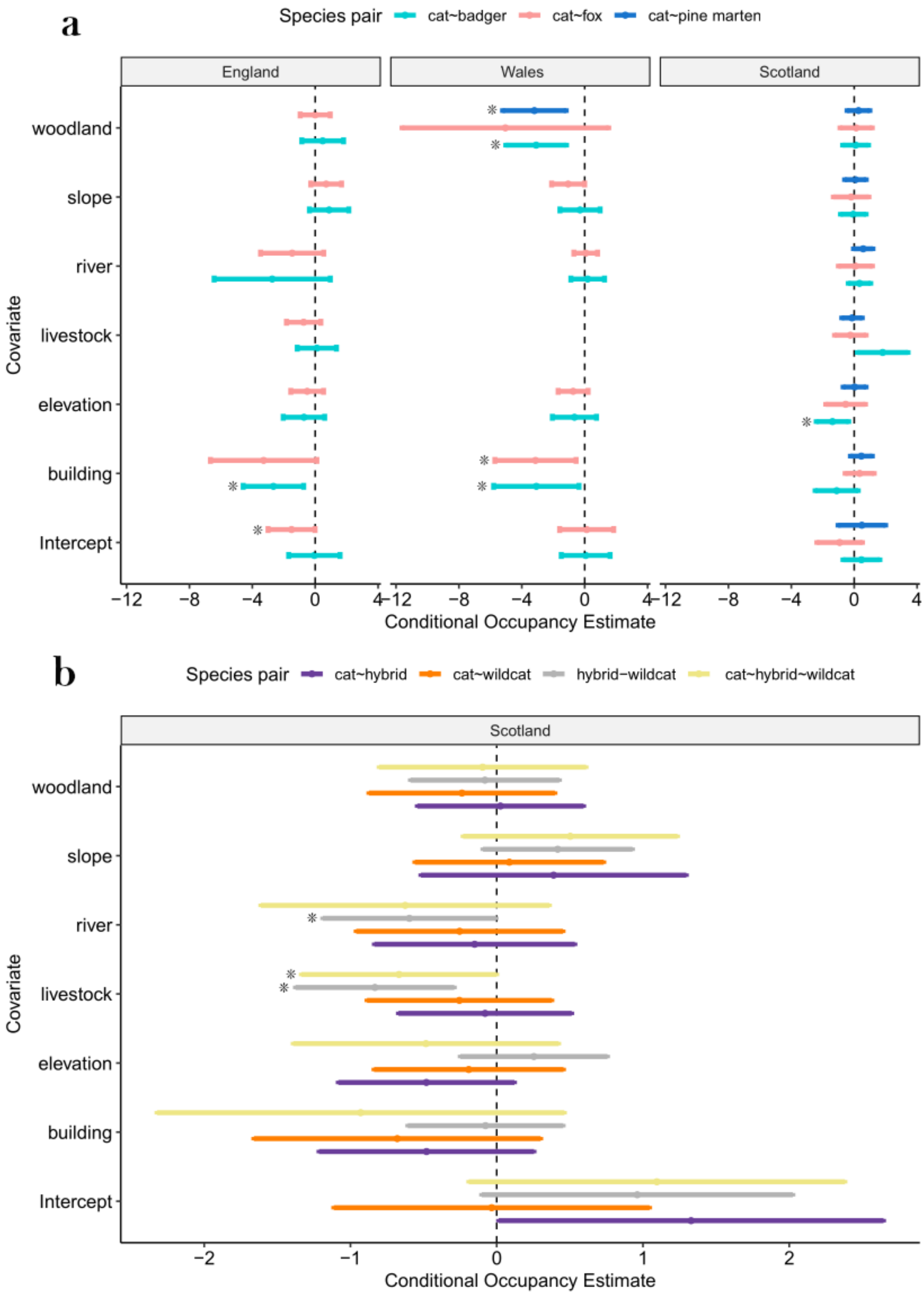

Figure 6.4 Comparison of multispecies parameter estimates for conditional occupancy probability *(ψ)* of domestic cats under sampled covariates conditional on the presence and absence of a) wild mesocarnivores and b) hybrid and wildcats, with 95% credible intervals. * Indicates coefficients with strong relationships which do not overlap 0

Temporal activity

Domestic cats had the highest temporal overlap with wildcats and hybrids in Scotland with all three species most active at dusk but with some activity recorded throughout the 24hr period (Figure 6.5). Conversely, domestic cats and badgers displayed significant levels of temporal separation at all sites. Badgers exhibited consistent nocturnal patterns of activity, while cats displayed daytime activity to varying degrees at all sites, with greater diurnal activity in England compared with Scotland and Wales. Domestic cats also had a high overlap with foxes in both Scotland and Wales (Figure 6.5). Overlap between domestic cats and pine martens was similar in Scotland and Wales (Figure 6.5) albeit with large confidence intervals in Wales owing to the small number of pine martens recorded during the study. This is a consequence of the population in Wales being new and increasing, with the study area being at the edge of its range at the time of data collection.

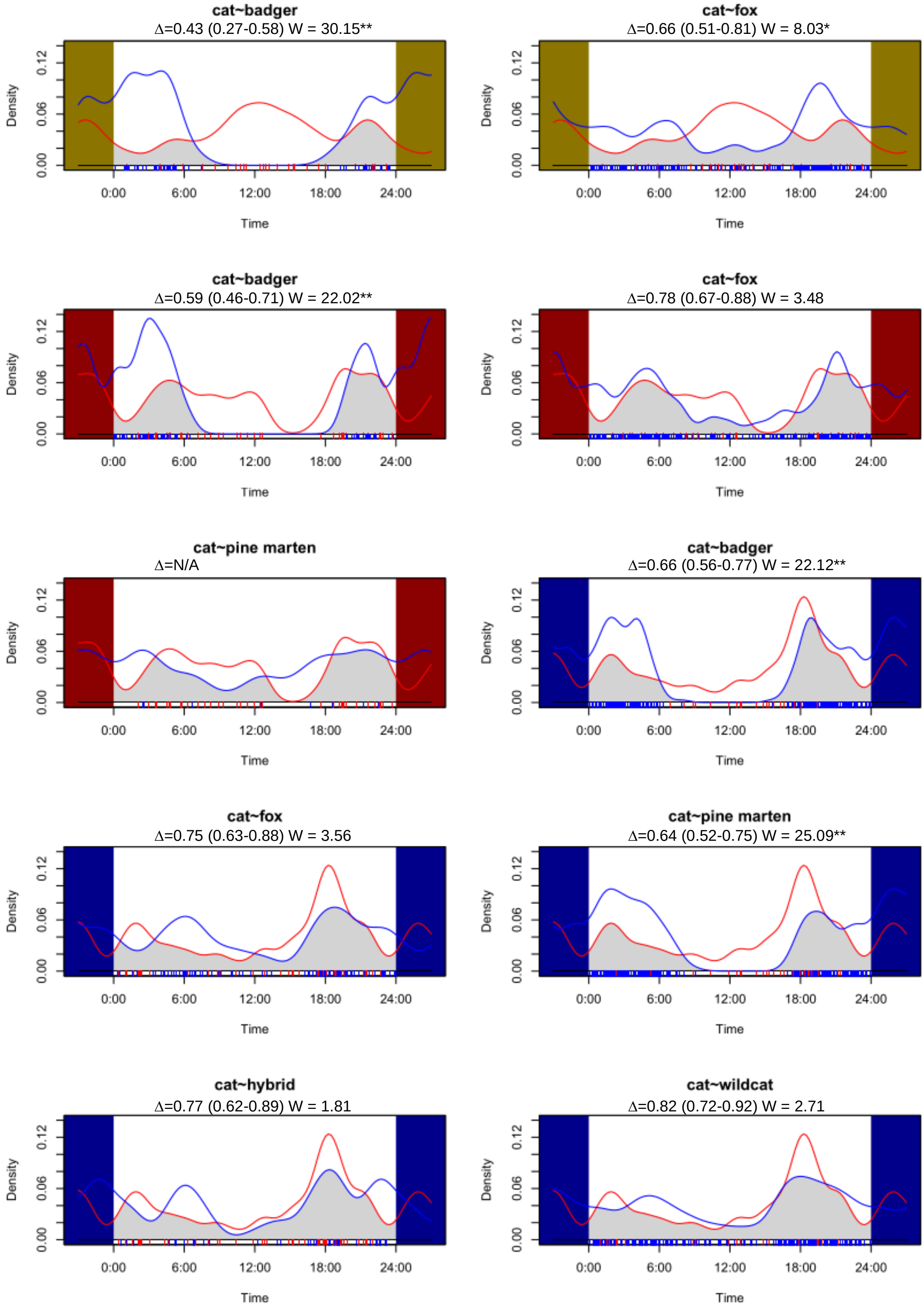

Figure 6.5 Temporal overlap plots of domestic cat (red line) and wild mesocarnivore (blue line) pairs (solid line) for each site (England = Yellow, Wales = Red, Scotland = Blue) Gray areas underneath density curves represent the overlap coefficient Δ. N/A is present when the number of mesocarnivore species detections is ≥10. *p < 0.05, significant; ** p < 0.01, highly significant

Discussion

Although numerous studies have sought to discuss the impact and management of cats and others have used camera traps to assess interactions among wild species, few have sought to address how the natural mechanisms of control exerted on cats by wild competitors influence their behaviour. Our study confirms the influence of these relationships and further, shows that habitat can modify the interspecific interactions between cats and wild mesocarnivores. Human settlements, agriculture and temporal activity play a significant role in influencing coexistence and competition. Moreover, the occupancy of mesocarnivores in the landscape is shown to influence the spatial mobility of cats. Our results, therefore, have implications for the management of wild mesocarnivores and hybridisation between cats and wildcats in variable human-influenced landscapes.

Cats and wild mesocarnivores

Domestic cats are predominantly associated with human settlements (Ferreira et al., 2011; Wang et al., 2015) and their introduction into the environment can intensify sources of competition for wild mesocarnivores surrounding areas of human activity (Cypher et al., 2017). At sites in England and Wales, domestic cat occupancy was primarily, and unsurprisingly, explained by their proximity to buildings, however, our results also suggest complexity in predicting the effect of habitat on how domestic cats use their environment. In Wales, domestic cats occur in areas of less than 50% woodland cover, whereas in England and Scotland, occupancy remained constant as woodland cover changed. In addition, we see that building distance does not significantly affect cat occupancy at our Scottish sites. For Scotland, this pattern may be masked by the cryptic nature of hybridisation: some of the apparent ‘domestic’ cats will be genetic hybrids (Kitchener and Senn 2023). Genetic data in Scotland obtained mainly from live trapping, including at some from the site used here, have found that there is a positive correlation between their proportion of wildcat genetic ancestry and the distance from inhabited buildings the cats were captured, with no cats scoring as domestic cat captured further than 229m from an inhabited building (Campbell et al. 2023c). However, this pattern of landscape use is complicated by presumably substantial interactions (e.g. by territoriality) between the different types of cats in Scotland with some indication of strong (though variable and not significant) negative effects on occupancy with building distance based on morphological distinctions alone. This variation in responses between sites implies that in some landscapes, proximity to human habitation provides only a basic indicator of spatial patterns of domestic cat activity and that both habitat characteristics and the presence of wild mesocarnivores mediate how cats use and therefore impact landscapes. This is shown by the relationship between domestic cats and badgers and foxes. In England and Wales, the probability of domestic cats occupying sites away from buildings declined when badgers or foxes were present compared to when they were absent. The probability of domestic cats occupying high elevations and high livestock holding density in Scotland was also reduced when badgers were present in comparison to when they were absent. This indicates that fluctuations in fox and badger populations, whether natural or because of human activity, could alter domestic cat mobility and in some circumstances facilitate the negative effects of domestic cats on wildlife away from human habitation.

The occupancy probability of foxes was considerably higher than any other mesocarnivore in both England and Wales, while in Scotland this was not the case. The lower occupancy probability of foxes in Scotland is mirrored by National Gamebag Census (NGC; Aebischer et al., 2011) data which shows a reduction in records of foxes killed on game estates in Scotland since 1995 suggesting a declining population. Hunting pressure is a primary cause of fox decline historically, while contemporary trends correlate with declines in rabbit populations (Sainsbury et al., 2019). We can surmise that disparity in these pressures within our study sites may explain the range of fox occupancy probability between them and subsequently the reduced influence of foxes on the spatial behaviours of cats in Scotland. Cryptic hybridisation in phenotypical ‘domestic’ cats could further influence occupancy, with genetic hybrids more likely to range further from human habitations (Campbell et al., 2023c). In addition, increased competition for resources resulting from an increased diversity of mesocarnivores in Scotland compared with England and Wales may play a role in the ranging behaviours of domestic cats.

Rather than the diversity of mesocarnivores influencing cat occupancy, it appears that specific species are more important in exerting control over cat mobility than others. Wild mesocarnivore control can influence the abundance, density, resource use and interactions of other mesocarnivores. The removal of foxes has also been shown to alter cat resource use and potentially population size (Molsher et al., 2017), while their presence can reduce interactions between cats and wildcats (Gil-Sánchez et al., 2015; Rodríguez et al., 2020). In a cat-wildcat context, this may suggest that historic and/or present-day control of foxes and badgers may increase opportunities for interactions between cats and wildcats. Thereby increasing the risk of genetic introgression.

Domestic cats, hybrid cats and wildcats

The relationship between domestic cats, hybrids and wildcats was primarily influenced by livestock holding density, with greater co-occurrence between the three types in areas with reduced density and wildcat occupancy negatively associated with increased holdings of sheep, poultry, and game birds. Wildcat use of agricultural land has been shown to reduce with increased agricultural intensity in mainland Europe (Ruiz-Villar et al., 2023) and our finding would imply the same is true in Scotland, though wildcats have been found to make use of farmland both in Scotland (Kilshaw et al., 2023) and elsewhere in Europe (e.g., Jerosch et al., 2018). Farms have been highlighted as a potential sink for hybridisation, due to the availability of small mammals and rabbits around farmland acting as a lure for wildcats to areas favoured by feral and domestic cats (Germain et al., 2008). Our analysis suggests that the type of agricultural activity as well as its intensity may play a role in influencing the extent of interactions, and therefore hybridisation risk, with spatial overlap found to be rare around types of livestock holdings which are perceived to have an increased likelihood of conflict with mesocarnivores (sheep, poultry, gamebirds).

Significant levels of spatiotemporal overlap between domestic cats and hybrids are also found as indicated by Kilshaw et al (2015), suggesting similar habitat requirements. Consequently, in situations where they share space, the potential for interactions is likely to be high. Strong temporal overlap relevant to the habitat between domestic cats and wildcats, as well as other wild cat species, has been shown in other studies (e.g. Daniels et al., 2001; Germain et al., 2008; Lynam et al., 2013; Soyumert, 2020). While domestic cats and wildcats had strong temporal overlap, spatial co-occurrence was less common, nor influenced by any of our covariates. In comparison, spatial co-occurrence between hybrids and wildcats was influenced by both livestock holding density and river distance, with greater overlap when closer to rivers and at lower livestock holding densities. This paints a picture of a spatial continuum, where animals existing on a spectrum of hybridisation share space with both domestic cats and wildcats, which can be spatially segregated in certain environments (Germain et al., 2008; Kilshaw et al., 2016). We must acknowledge some uncertainty due to the potential for misclassifying cats, hybrids and wildcats when using pelage scoring, which may influence these findings (Senn et al., 2019). It would be valuable to conduct a more in-depth study using only genetically assessed cats across a wider spatial extent, to reassess the influence this plays on interactions between cats, hybrids and wildcats.

Management implications

Our study suggests that partitioning between cats and wild mesocarnivores is affected by habitat, temporal behaviours, and interspecific interactions. The spatial activity of cats is likely to be affected by population fluctuations of wild mesocarnivores, as well as the absence of top-order predators. This presents a management conundrum. If we accept the premise that as introduced animals living at high densities, cats have negative impacts on their environment, then the potential for wild mesocarnivores to reduce the spatial extent of cat impacts is broadly a positive one. However, many wild mesocarnivores are frequently in perceived or actual conflict with rural industries and conservation activities and are controlled (Dickman, 2010; Cassidy, 2012; Macdonald and Johnson, 2015; Baines et al., 2022). Control of mesocarnivores is viewed by some as keeping their population in balance in the absence of top-order predators in Great Britain. Furthermore, control is sometimes viewed as beneficial to the conservation of other species (Baines et al., 2022) and farming activities (Macdonald and Johnson, 2015). It is, therefore, possible that control of wild mesocarnivores may have unintended consequences on the accessibility of the landscape for free-roaming cats (both domestic and feral).

In the context of wildcat conservation, and in particular reintroductions, an abundant and diverse mesocarnivore community may provide a buffer which reduces interactions between cats and wildcats (Gil-Sánchez et al., 2015). Thereby, creating a greater degree of self-regulation of populations within mesocarnivore communities (Curveira-Santos et al., 2021), which would generally be considered desirable. The caveat to this is that wild mesocarnivores also represent a threat to wildcat kittens (Götz et al., 2008), in a reintroduction context selecting sites where mesocarnivores are abundant could therefore lead to increased juvenile mortality among wildcats and hinder their establishment.

In Scotland, which had the most complete mesocarnivore guild of our sites, domestic cats and wildcats have limited spatial overlap, but unsurprisingly there was greater overlap by both species with hybrids. Focusing on reintroduction and conservation initiatives for wildcats in areas where the number of hybrid animals is low or absent is therefore likely to reduce opportunities for introgression between cats and wildcats (sensu Biro et al's 2009 'hybrid bridge' hypothesis). However, relatively rare encounters are all that might be required, given the mating system of wildcats, for hybridisation to occur. Therefore, this risk would need to be further reduced through the neutering of free-roaming cats, although effective schemes to do this are not easy to achieve, with studies indicating that neutering programmes would need to neuter 75% of the unneutered population to be effective (Andersen et al., 2004). Nevertheless, neutering programmes in the presence of hybrids may be even more difficult (Campbell et al. 2023b), illustrating the benefits of investment in such programmes at the outset.

Cats are frequently portrayed as being highly abundant in Britain and while that may be true in urban areas (Aegerter et al., 2017; McDonald and Skillings, 2021), in our study they had the lowest occupancy probability compared to wild mesocarnivores present at every site. Furthermore, we find evidence that the spatial extent of cat impact in rural areas is low beyond a couple of hundred meters from human habitation. In terms of cat management, this suggests that the number of free-roaming feral animals is likely to be small and management will be most effective around settlements. This does not mean that isolated colonies do not exist, and seasonal effects may increase their roaming, and consequently their impact, but it does suggest that in rural landscapes cat populations are spatially limited. Combined with our other results we suggest that consideration of the effects of mesocarnivore control and species restoration on the ecological impact of cats should be an important component of discussions around the management of rural cats and restoration of wildcats.

Chapter Seven: Discussion

The restoration of species is a priority in global conservation (Seddon et al., 2014). Increasingly, species restoration is also at the forefront of public debate, conservation, and policy around the future of British landscapes and seascapes (Carter et al., 2017; Sandom et al., 2019). In this thesis, I aimed to explore and inform the social and ecological feasibility of reintroducing the European wildcat to England and Wales by defining best practices in the assessment of social feasibility (**Chapter Two**) and exploring the perspectives of key stakeholders toward a wildcat reintroduction (**Chapters Three & Four**). Moreover, I identified potential release areas and release sites for wildcats in England and Wales (**Chapter Five**) and examined the presence and space use of domestic cats in the context of interactions with wild mesocarnivores and hybridisation with wildcats (**Chapter Six**). Here, I synthesise the work in this thesis to highlight key contributions toward scientific knowledge and key findings for wildcat reintroduction related to both social and ecological dimensions and place these into the discussion around species reintroduction, restoration, and rewilding more widely. Finally, I bring this together to form a series of recommendations for practitioners interested in the reintroduction of wildcats in England and Wales.

Key scientific and social scientific contributions

In **Chapter Two,** I conduct the first comprehensive review of if, how and when social feasibility assessments are conducted. The review highlights the importance of such assessments but also that they are not conducted by the majority of reviewed projects. I also describe how a lack of capacity and resources for social sciences in conservation, and failures to record experiences and share best practices, are barriers to effective social feasibility assessment.

In **Chapter Three,** I emphasise the influence of wider discourses around modern conservation practices and the media depiction of farmers and conservation on farmer perceptions of projects. This research suggests there is value in taking a broader approach to engagement and research to understand the underlying reasons for stakeholder views, which are often not to do with a specific species.

In **Chapter Four,** I highlight the issue of unclear lines of responsibility for liminal cats and how these influences discussions of the management, impacts and welfare of unowned cats. This research fills an important gap in knowledge by linking these findings to the conservation and restoration of European wildcats.

In **Chapter Five,** I undertake the first analysis of habitat networks for wildcats in in England and Wales and highlight the value of mixed method approaches to decision-making. This makes an important contribution to the literature by highlighting the intertwined nature of social and ecological factors to decision-making in the context of reintroductions, specifically around where they should occur.

In **Chapter Six,** I deduce that the spatial ecology of domestic cats is influenced by habitat characteristics and the presence of certain wild mesocarnivores. Moreover, I hypothesise that the restoration or decline of wild mesocarnivores will influence the impact of domestic cats concerning predation, disease transmission and hybridisation with wildcats.

My findings expand our understanding of important social elements of species reintroductions and wider conservation initiatives, concerning how social feasibility assessments are conducted, and the influence of stakeholder perceptions on the framing of and approaches to engagement, as well as management interventions. I highlight the need for further social science research around reintroductions and other conservation practices to enable the sharing of ideas and ongoing development of best practices. Above all, this thesis shows the need for research around reintroductions to be truly multi-disciplinary if they are to attempt to reconcile the needs of people and wildlife.

Social dimensions of a wildcat reintroduction

Social Feasibility

Species reintroductions, whether to support conservation, ecosystem function, rewilding, or all the above, should assess social feasibility before they proceed (IUCN/SSC, 2013). In this thesis, I find assessments of social feasibility to be completed by a minority of reintroduction projects, and when undertaken are often insufficient in scope to be able to accurately determine if a project was socially feasible (**Chapter Two**). This builds on work by Seddon et al (2007) which suggested only 4% of reintroduction literature included social dimensions and work by Bennett et al (2017b), which emphasises the need for greater uptake of the social sciences in conservation. Recognition of the importance of conservation social sciences has grown significantly in recent decades (Bennett et al., 2017a). Despite increased discussion around the importance of studious early examination of the human dimensions of reintroduction and other conservation projects (Ban et al., 2013; Bennett et al., 2017b; Bennett & Roth, 2019; Bubac et al., 2019; Sandbrook et al., 2013; Sterling et al., 2017), including within the IUCN guidelines (IUCN/SSC, 2013), this appears to remain understudied or poorly implemented in a reintroduction context. **Chapter Two** represents an important initial exploration into current practices in the assessment of social feasibility and can be used to further develop the IUCN guidelines, as well as incentivise and inform the delivery of social feasibility assessment in translocation projects.

Chapter Two also explores barriers to social feasibility assessments and finds similarly to other conservation practices, that a failure to prioritise resources to human dimensions and a lack of expertise in social sciences during planning are significant barriers (Ban et al., 2013; Sanborn & Jung, 2021). The inclusion of conservation social sciences in planning can improve our understanding and decision-making at varying scales, from better understanding the motivations of individuals, to identifying concerns affecting local communities, to wider discussion around species restoration at the national level (Bennett et al., 2017a). Moreover, it can help to identify the cultural appropriateness of decisions and the framing of potentially sensitive projects (Bennett et al., 2017a). In the context of a wildcat reintroduction to England and Wales from where the species has been absent for a long period and where its reintroduction may have cultural connotations, I suggest that the inclusion of social scientists and those with local knowledge during the planning phase will enhance the implementation of a robust and bespoke assessment of social feasibility. Importantly, **Chapter Two** also suggests that doing so is likely to increase the chances of a successful reintroduction.

Stakeholder engagement

To date, there is little published information on stakeholder attitudes and understanding of the wildcat throughout its range. The results in **Chapter Two** around the need to prioritise social dimensions of wildcat reintroduction are further highlighted in **Chapters Three** and **Four.** These chapters show that before reintroduction to England and/or Wales can be considered socially feasible, the wildcat needs to be restored to the social and cultural landscape. Current low knowledge levels are a significant hindrance to communicating nuanced information about wildcat ecology and the potential impacts of a reintroduction (Hiroyasu et al., 2019). These results are in keeping with expectations of salience declining with increased time since extinction (Jarić et al., 2022), with the wildcat being absent from England and Wales for ~150 years. More broadly, as a species which is extinct or threatened in many parts of Europe (European Commission, 2015; Gerngross et al., 2021) and one which is largely elusive, it is likely that without conservation messaging, a lack of interest in or ignorance toward wildcats and their conservation could impact support for conservation efforts throughout Europe (Vincenot et al., 2015).

My conclusions in **Chapter Two,** that early engagement and long-term commitments to people and places are more likely to result in successful outcomes, were echoed by farmers in **Chapter Three**. Such commitments and engagement will make it easier for conservation organisations to act as a trusted source of information. Familiarity, face-to-face interactions, and cultural sensitivity were all valued highly by farmers in fostering positive engagement. The benefit of early engagement is of particular importance to a wildcat reintroduction. First, early engagement will help conservation practitioners identify and address concerns and potential areas of conflict before they occur, enabling solutions to be discussed in a timely manner (Redpath et al., 2013; Reed, 2008). A good example of this is in the context of South West England where high densities of gamebirds and poultry holdings indicate a risk of conflict with rural industries. As such, early engagement with these groups will be key to conflict mitigation. Second, **Chapters Three** and **Four** highlight uncertainty and misrepresentation of the potential benefits and impacts of wildcat reintroduction. Early engagement and outreach will be essential to address this, as lingering misconceptions may have a significant influence on expectations and ultimately on support or opposition (Consorte-McCrea et al., 2022; Gusset et al., 2008; Hiroyasu et al., 2019). Third, farmers interviewed in **Chapter Three** perceive conservation messaging in the media and social media to be problematic, this may require conservationists to prioritise resources for sustaining a visible and accessible presence within affected landscapes early in the process. I also found significant individuals (in the media or conservation) or terms (such as rewilding in Wales), that are perceived to be divisive, impacting support by association. Clearly defining and communicating who is responsible for delivering the project early on, and what its ultimate goals are, is therefore important.

Domestic cat management

Hybridisation between wildcats and domestic cats is frequently studied in the context of its genetic (Tiesmeyer et al., 2020), ecological (Germain et al., 2008) and conservation (Breitenmoser et al., 2019) consequences. However, the root cause of hybridisation - the presence of unneutered domestic cats in the environment - is social as much as it is ecological (Crowley et al., 2020a; Slater & Shain, 2005). A better understanding of the social dimensions of domestic cat management is therefore critical to informing effective solutions. In **Chapter Four,** few cat owners knew of hybridisation as a threat, nor about the role cat owners can play in reducing it. There is a clear delineation between awareness of the species in Scotland compared with England and Wales. In Scotland, the wildcat is extant, the subject of extensive conservation action and communications as well as of symbolic and cultural value, with cat owners who are aware of key issues (Bacon, 2017; Campbell., et al., 2023; Wemyss et al., 2023; Williams Foley, 2022). This thesis suggests the wildcat is largely forgotten by our sampled stakeholders and, where remembered, is often associated with Scotland and the ‘Scottish wildcat’ name. Both farmers and cat owners as owners or hosts of cats are critical stakeholders in mitigating hybridisation. Consequently, the first stage in any proposed English or Welsh reintroduction is the re-establishment of the wildcat as a native British species in the public consciousness. The positive influence of conservation actions on the awareness of wildcats and hybridisation in Scotland can provide both encouragement and a valuable source of information to practitioners working on the species elsewhere.

The management of domestic cats (owned, semi-owned and unowned) has been identified as fundamental to the successful long-term restoration of wildcats in Britain (Campbell., et al., 2023; Littlewood et al., 2014). Evidence from Scotland suggests that a bespoke long-term TNVR programme is needed to manage unowned cat populations, and effective communications are required to encourage awareness of the need to neuter and vaccinate owned cats in the context of wildcat conservation (Campbell., et al., 2023). The findings in **Chapter Four** highlight that in England and Wales, there is a need to bring together stakeholders to achieve effective cat management. Furthermore, collaboration with those involved with this issue in Scotland is likely desirable when discussing cat management in the context of hybridisation across Britain. Management of unowned cats is desirable from the perspective of sampled cat owners **(Chapter Four)** but the most appropriate methods, and who is responsible for delivering management actions, were debated. Such debate over the effectiveness, ethics, and achievability of cat management approaches, both legislative and practical, are prevalent in many studies (Boone et al., 2019; Crawford et al., 2019; Crowley et al., 2020a; Wolf & Schaffner, 2019). I propose that collaboration between a wide range of stakeholders can help bring together the diverse priorities and concerns of different actors and address the liminal position of unowned cats which we identify as a barrier to effective management. Management of unowned cats is already conducted by cat welfare groups, rescue centres and concerned individuals across much of Britain, albeit often in reaction to sightings or welfare concerns. Wildcat restoration throughout Britain could be viewed as a potentially useful catalyst to bring key stakeholders together and discuss a national plan for the management of domestic cats.

Together **Chapters Two, Three** and **Four**, provide a valuable understanding of the human aspects of wildlife translocations and wildcat reintroduction specifically. **Chapters Three** and **Four** expand upon the lessons and best practices around social feasibility set out in **Chapter Two** and begin to apply some of this learning to key stakeholders in a wildcat reintroduction context. Through the early exploration of stakeholder perceptions and concerns, I have set out where potential divisions and consensus may be found and detailed the importance of prioritising social research and effective engagement. Together these chapters provide a starting point with which to inform engagement, consultation, and outreach toward a robust assessment of social feasibility.

Ecological dimensions of a wildcat reintroduction

Selection of release areas and sites

Where wildcats might be reintroduced in England and Wales is a central question to answer before meaningful planning can be conducted. In **Chapter Five** I build on work by MacPherson et al (2020), the only published assessment of wildcat habitat in England and Wales, by exploring the viability of potential release areas. I find that West Wales represents the most likely landscape to host a viable wildcat population. South West England is also shown to have a sufficient quantity of connected habitat, however, risks around roads, habitat fragmentation, and interactions with domestic cats appear greater. Based on a combination of home range estimations from other studies, I infer under the most cautious modelled scenario a potential carrying capacity of 582 (413-985) wildcats in West Wales. The establishment of a minimum viable population of wildcats was estimated by Littlewood et al (2014) to require at least 40 individuals including 20 females to be viable in the long term. Moreover when estimating the short and long-term viability the 50/500 rule is often used, i.e. a minimum population size of 50 is necessary to combat inbreeding and a minimum of 500 individuals is needed to maintain evolutionary potential (Franklin, 1980; Pérez-Pereira et al., 2022). Using both measures the findings in **Chapter Five** suggest West Wales is the only studied region to have a carrying capacity able to meet both criteria under the worst-case scenario, albeit South West England is on the edge of these values at 495 (351-837).

Of all the assessed regions and candidate release sites, **Chapter Five** highlights an area centred within Carmarthenshire, West Wales, as providing the best opportunity to support an initial reintroduction. This area is similar to that which MacPherson et al (2020) identified. Optimal wildcat habitat types, such as broadleaf woodland and wooded riverine valley systems and woodland edges are complimented by small-scale conifer plantations. The proportion and proximity of these habitats facilitate connectivity between patches of agricultural land (Jerosch et al., 2018). This will assist the initial establishment of wildcats by making it easier for wildcats to disperse from release sites. Moreover, in

Chapter Six I find that within West Wales domestic cats are largely anchored to their home, with occupancy negatively affected by both woodland and the presence of other mesopredators, the presence of high-quality wildcat habitat should therefore reduce the risk of interactions. In addition, pine marten have successfully recolonised large areas since their translocation from Scotland (McNicol et al., 2020). This suggests that, while there are differences in ecology between the species, the landscape is permeable to a reintroduced mesopredator with a preference for wooded environments. In contrast, in South West England, domestic cats were found to travel farther from buildings and showed no avoidance of woodland.

The results in **Chapter Six** suggest that the presence of other wild mesocarnivores may provide a buffer that reduces spatial overlap and interactions between domestic cats and wildcats. This has been suggested elsewhere (Gil-Sánchez et al., 2015; Rodríguez et al., 2020) and could help mitigate hybridisation risk in a wildcat reintroduction context. However, mesocarnivores also represent a threat to wildcat kittens (Götz et al., 2008), so selecting reintroduction sites with abundant mesocarnivore populations could lead to increased juvenile mortality. There is likely a balance to be struck - choosing sites with established but not overly dense mesocarnivore guilds, while also focusing on areas with few or no hybrids cats present. An additional consideration will be if supplementary food is provided in a soft-release scenario as this may attract competitors and artificially inflate their densities surrounding release sites, potentially fuelling negative interactions. This is a particular concern if, as in Scotland, captive animals are used, as they will have little predator avoidance (Jule et al., 2008). The continued monitoring of mesocarnivores surrounding release sites before and after release is imperative to decision-making around release strategies. Moreover, ongoing population-level neutering of free-roaming domestic cats is a critical accompanying measure to further reduce opportunities for introgression between cats and reintroduced wildcats.

Threat management

The threat of introgression is widely cited as the most impactful to wildcat populations. Introgression risk was previously attributed primarily to unneutered feral domestic cats. This thesis however reinforces the hypothesis that interactions with hybrid cats may be more influential in hybridisation risk (Germain et al., 2008; Kilshaw et al., 2016). Hybrid cats in Scotland are found in **Chapter Six** to have significant spatiotemporal overlap with both wildcats and domestic cats, which are shown to be largely isolated from one another. In West Wales, results suggest that domestic cats are anchored to buildings and absent from woodlands, largely limiting potential interactions with reintroduced wildcats. In this instance hot spots for potential interactions between domestic cats and reintroduced wildcats are confined to rural buildings near or between patches of quality habitat, or that offer an abundance of food. However, in South West England, domestic cats are shown to roam further from buildings and into better quality wildcat habitat, moreover, in **Chapter Five** we see that South West England has the highest densities of domestic cats of any study region. This suggests an increased risk of introgression when compared to West Wales. It is important to note, that in both regions, domestic cats detected in **Chapter Six** were invariably owned and neutered, meaning the exact introgression risk is harder to quantify and will require longitudinal monitoring of candidate regions and collaboration with local cat owners, welfare organisations and rescue groups engaged in TNVR activities. Additionally, further modelling around hybridisation risk under different release scenarios should be undertaken.

Road mortality is the most common cause of human-induced mortality for wildcats (Bastianelli et al., 2021). In **Chapter Five**, road networks are found to significantly impact the size of potential release areas in West Wales and are the primary source of pinch points as well as causing a large reduction in available release sites in South West England. The barriers to dispersal created by higher road density, particularly multi-lane and high-traffic roads, can also reduce gene flow (Westekemper et al., 2021), hindering the establishment of a genetically diverse population. Consequently, taking steps to mitigate the potential for road mortality and improving the permeability at functionally important crossings is likely to be critical to the connectivity of each region and the reduction of road mortality post-release. Mitigation measures like road underpasses or overpasses, complimented by wildlife fencing, can improve road permeability and facilitate safer crossings for species like the wildcat (Klar et al., 2009; Kramer-Schadt et al., 2004). Where present these are often focused on singular large roads rather than considering the density of roads in an area which has been shown to be just as impactful (Westekemper et al., 2021). Roads are explicitly considered within corridor mapping, in addition, distance to major roads and road density are both considered when selecting candidate release sites. Therefore, the results should be robust in supporting the initial release of wildcats. Nevertheless, It is advisable to further consider potential barriers to long-term dispersal along likely dispersal routes, roadkill data is frequently used in wildcat research, while data on roadkill hotspots for other similar-sized species is also available and could be considered in future planning.

As in Scotland, the landscapes of England and Wales are working landscapes. Management practices for forestry wind farms, game, and agriculture will therefore impact the quality of habitat for wildcats, as well as potential threats (Simon and Lang 2014; Campbell et al., 2023d). Subsequently, liaising with these groups to raise awareness of returning wildcats and how this relates to management practices is required. The development of best practices for relevant industries is ongoing in Scotland, with discussions around but not limited to; reducing the use of rodenticides in agriculture and mainlining areas of scrub on farmland; the creation and monitoring of brash and log piles; and an awareness of the wildcat breeding season and the risk of disturbance in forestry (Campbell et al., 2023d). Outcomes from these discussions are likely to be directly relevant to practices in England and Wales. In all modelled region, forestry plantation is identified as important for connectivity and as potential release sites, while in South West England high densities of game and poultry holdings are identified.

Many of the issues finding a suitable ecological landscape revolve around social aspects, whether that’s the attitudes and practices of key stakeholders and the likelihood of wildcats interacting with them; the role of cat management and cat owners; or the political decision-making over making roads safer for wildlife and people. In the longer term, we also would highlight the role of local involvement, through volunteers, citizen scientists, local decision-makers, and community ownership in making a project viable. This emphasises the value of taking a multi and interdisciplinary approach to the species reintroduction process as I have here, which explicitly acknowledges the intertwined nature of the social and ecological aspects of a wildcat reintroduction.

Implications for reintroductions, restoration, and rewilding in Britain

This thesis highlights the need to better integrate social and ecological factors in conservation and manage the connections between domestic and wild systems. Effective human engagement combined with ecological knowledge can achieve better conservation outcomes in these complex socio-ecological contexts. Many if not most projects have social factors which affect ecological outcomes, and ecological factors which affect social perspectives. It is, therefore, necessary to treat the social and ecological as interlinked and approach decision-making as such. The approach taken throughout this thesis and most evidently in **Chapter Five** to integrate social and ecological methods in decision-making is something that can be replicated across wildlife conservation programmes. Such an approach lays the foundations for additional ground-truthing by guiding decisions and emphasising areas of concern before in-depth consultation and habitat assessment being conducted.

Using the case study of the wildcat, the work here contributes to the growing body of research around the implementation of species restoration and rewilding in Britain. First, we identify in **Chapter Three** that the framing of projects and terminology can generate significant local reactions, including emotive responses. Moreover, the work in **Chapter Two** on assessing social feasibility in translocations is likely to be just as applicable to other conservation actions. Rewilding continues to gain traction as an approach to ecological restoration (Carver et al., 2021; Sandom & Wynne-Jones, 2019), however this thesis shows that in communities where rewilding projects have previously been attempted and taken a poorly perceived approach to social integration and engagement, e.g. west and mid-Wales (Holmes et al., 2022; Jones, 2022), there appears to be some remaining animosity to the detriment of future conservation actions. Furthermore, in regions like these which have been subjected to media interest (Monbiot, 2014), rewilding is symbolic of an anti-farmer narrative. This emphasises the need to understand not just current attitudes, but also the historical context of any areas where reintroductions, restoration or rewilding are being attempted, to tailor social interventions accordingly. In this context, while greater evaluation and transparency around success and failure in delivering social aspects of restoration projects is needed to further develop guidance, it is also true that cultures, histories, and economics, shift at a fine scale, and all influence the best approach to delivering effective social actions in wildlife conservation. Consequently, in some senses, every project needs to take a bespoke approach.

I show that the restoration of species and composition of species within an environment can influence interactions and mobility in relation to habitat use **(Chapter Six)**. Domestic cats have a large global population, and their hunting behaviours can threaten biodiversity, especially birds. Consequently, the role of competitive wild mesocarnivores in affecting domestic cat mobility is of interest beyond wildcats and to biodiversity more broadly. Restoring a diverse mesocarnivore assemblage may limit negative interactions. Furthermore, an understanding of how the restoration and management of mesocarnivores impacts both wild and domestic species is important to conservation decision-making.

Future directions and recommendations

Based on the assessed regions, there appear to be promising options for wildcat reintroduction in England and Wales. However, continued assessments incorporating additional stakeholders, evaluating site-specific conditions, and monitoring post-release outcomes would further develop the evidence base.

European wildcat conservation presents a somewhat unique challenge to practitioners. Literature increasingly points to wildcats being a highly adaptable species. This would suggest it is an ideal candidate for reintroduction. However, while most aspects of wildcat ecology are well-researched, our understanding of some important aspects remains incomplete. Specifically, the drivers of hybridisation, and understanding of wildcat reproduction in the wild, would greatly enhance confidence in reintroduction planning. Moreover, social attitudes, and how they link to potential persecution and species connection, require further research. For all these topics a trial reintroduction presents an ideal scenario to monitor, learn and engage to the benefit of wildcat science and conservation.

Recommendations for the reintroduction of the European wildcat

- *Conduct a robust and bespoke social feasibility assessment, with an emphasis on making long-term commitments to people, places, and partners*: The findings of this thesis provide insight into conducting a social feasibility assessment and the value of bespoke assessments and commitments among collaborators. This thesis informs the framing and implementation of engagement by highlighting key issues, opportunities, and approaches, but does not provide a representative sample of all stakeholder views within candidate landscapes. As such I recommend using this work as the basis for developing the next stage of engagement and consultation.

- *Restore the wildcat to the cultural and social landscape of England and Wales*: The lack of salience around wildcats requires practitioners to go above and beyond a local consultation, and to prioritise awareness raising of the history, ecology, and potential future of the wildcat in England and Wales at a regional and/or national level. This should be part of efforts to improve connection with the species and restore the wildcat as a British species.
- *Build a coalition of partners in England, Wales, and Scotland around the management of domestic cats*: I highlight that there is an opportunity for any reintroduction initiative to utilise the affiliation of cat owners toward all types of cats in building relationships with those who are natural allies of a wildcat reintroduction. These relationships are key to developing buy-in for responsible ownership practices around potential release sites as well as a potential source of support. Additionally, building relationships and collaborating with cat welfare groups and charities - in particular, those who already conduct Trap Neuter Vaccinate Release programmes - is essential both in determining the current situation regarding the presence of unneutered cats in rural landscapes, as well as the long-term mitigation of hybridisation.
- *Early engagement with farming communities in the proximity of release sites*: Engagement with farming communities likely to be affected by a wildcat reintroduction should begin at the earliest possible opportunity. This thesis suggests that this should involve face-to-face discussions as well as having a long-term visible presence within the community to build relationships.
- *Early engagement with gamekeepers and poultry farmers to discuss wildcat ecology and potential conflict mitigation*: The densities of game and poultry holdings have the potential to be a source of conflict in South West England. Therefore, I recommend early engagement and a process of dialogue to

build trust, awareness of wildcats and mitigation strategies that are amenable to all parties.

- *Be specific and transparent about the scope of the work being attempted*: ‘Rewilding’ is widely viewed as toxic within our sample and messaging is perceived to be anti-farmer by a small number of proponents in the media. Consequently, it will be important to frame the project carefully during the consultation, being specific and transparent about the scope of the work being attempted. Moreover, in publicising any reintroduction project in this landscape, conservationists need to consider the potential impacts of using certain public figures to promote a reintroduction.
- *Be sensitive to and integrate with Welsh culture and language*: The study region in Wales has a strong cultural identity, which in many cases is linked to the Welsh language and sense of history and place. Consequently, any project working in this region needs to authentically integrate these facets into its programme. Ideally, the involvement of local conservation groups and interested individuals from the outset will be important in this integration. Furthermore, practitioners should seek to spend time in the landscape to understand cultural and civic norms, use the Welsh language in communications and engagement, create a forum to include local people in decision making and be a visible and accessible presence within the landscape.
- *Continued monitoring of mesocarnivores, domestic cats and prey species around release sites*: Monitoring activities should occur for as long as practically possible. Data from camera traps are a valuable tool in identifying domestic cats, as well as monitoring long-term changes that occur as a consequence of a wildcat reintroduction. All of these are important in planning and post-release monitoring.

Concluding remarks

The continuing decline in global biodiversity has compelled significant conservation action. The reintroduction of species to reverse localised extinction and drive the restoration of ecosystems provides optimism but can also be a source of trepidation for those impacted by such endeavours. Consequently, a multi and inter-disciplinary approach to restoration is often needed. In this thesis, I promote an integrated social-ecological approach to conducting the reintroduction planning and feasibility process through an exploration of the social and ecological feasibility of reintroducing the European wildcat to England and Wales. I provide an understanding of what best practice might look like for social feasibility assessments and begin to establish how key stakeholders toward a wildcat reintroduction perceive wildlife conservation, wildcats and their role within both. This elucidates findings applicable to practitioners interested in or necessitated to engage with stakeholders, and how wildlife conservation practices might be adapted to be more effective at meeting social challenges. Finally, I evidence where and begin to discuss how, a wildcat reintroduction might be best suited to occur in England and Wales and demonstrate the role of competitive species and habitat characteristics in predicting the spatial ecology of domestic cats and their interactions with wildcats. Ultimately this research advances a holistic foundation to guide decision-making on the feasibility of wildcat reintroduction. It provides recommendations for conducting social feasibility assessments, stakeholder engagement, release site selection, threat management, project scoping, and integrating cultural aspects that could enable the return of wildcats to the English and Welsh countryside. The interdisciplinary perspective applied here serves as a model for comprehensive planning essential to successful conservation translocation.

Appendices

Appendix 1: Methodological details of the social feasibility review process and outputs associated with Box 1 (Chapter Two)

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Systematic review methodology

We conducted a review of conservation translocation projects, including case studies identified as a Population Restoration; Reinforcement; Reintroduction; Conservation Introduction; Assisted Colonisation; or Ecological Replacement, as defined by the IUCN. We initially reviewed the IUCN Global Reintroduction Perspectives book series, which returned 416 suitable case studies. Visiting IUCN specialist group websites enabled the identification of a further 18 action plans, proposals and reports of translocation case studies not published in academic journals. Finally, we identified further case studies through literature searches using Google Scholar, where Boolean search terms 'reintro*' 'translocat*' 'reinforce*', 'restor*', and their synonyms were used, in conjunction with taxonomic groups 'mammal' 'amphibian' 'reptile' 'invert*' 'fish' 'bird' 'plant' and the term 'feasibility'. The literature searches returned 1,451,761 results across all searches. The results were filtered for each search by selecting only the 500 papers deemed most relevant by Google Scholar, meaning 14,000 papers were screened. We excluded all studies, which primarily focused on species ecology, before identifying whether the study's subject fitted one or more IUCN definitions of conservation translocations. Duplicate case studies were removed. This process returned a further 116 case studies meeting our selection criteria. In total, 550 case studies met the criteria for inclusion.

The use of both Google Scholar and Web of Science was piloted before the final searches were conducted. Both performed very similarly with regards to identifying literature in academic journals, however, Google Scholar was able to identify greater numbers of studies in the 'grey literature'. While Google Scholar has limitations with replication, the need to include grey literature was felt to be important to make the sample as representative as possible, therefore it was deemed to meet the needs of the project better. We believe this approach provides a comprehensive review of works published in scientific journals and by the IUCN, as well as accessible grey literature, enabling a meaningful exploration of this subject, while acknowledging that a greater focus on abandoned projects, unpublished work and non-IUCN associated materials would be desirable in further, primary studies.

All case studies were systematically coded in NVivo v12 by a single coder (TRD). Data collected from each case study included: the project start year, focal species and taxon, and the IUCN region where the project took place. From each case study, we extracted statements describing actions relevant to one or more of the IUCN social feasibility guidelines (Table 2.1). These statements were cross-coded within categories pertaining to the ten social feasibility guidelines, the project stage that a relevant activity was undertaken ('Feasibility', 'Implementation', and 'Post-release'), and whether statements were described as 'Reasons for failure', 'Difficulties faced', or 'Reasons for success'. We used the framework from the IUCN Reintroduction Perspectives publications to define the project stage. Statements coded into 'Reasons for failure', 'Difficulties faced', or 'Reasons for success' sections were then thematically sub-coded into groups based on the similarity of the described activity (Table S1.1).

We analysed simple trends in the number of projects evidencing the inclusion of one or more of the social feasibility guidelines concerning the timing of the publication of the Guidelines (Pre-Guidelines, post-1998 and post-2013), IUCN regions, and taxonomic groups. We used χ^2 tests of association and, where significant, adjusted residuals were derived to indicate which category had the greatest influence. To analyse the frequency that each of the social feasibility guidelines was evidenced at the Feasibility stage, we used χ^2 tests of variance. Count data were used to assess the number of social feasibility guidelines that each case study evidenced at the Feasibility stage. To identify common themes relating to 'Reasons for failure' and 'Reasons for success', we used Jaccard's similarity coefficient, to ascertain similarity between pairs of grouped statements. A value of 0 indicate no similarity between the two groups, whereas a value of 1 indicates a strong similarity. Where a score between 0.5-1 (highly similar) was returned the sub-categories were merged; this continued until we had distinct, dissimilar groups relating to ‘Reasons for failure’ and ‘Reasons for success’.

Table S1.1. Most frequently described Reasons for success and Reasons for failure associated with the themes of the social feasibility guidelines

Reason for success	**Case studies (n=177)**	**Reason description**
Stakeholder organisation	40	Projects conducted detailed planning and agreements between stakeholders. They made long-term commitments and support at local, national and international levels to increase resource capacity and resilience, helping to spread risk.
Engagement mechanisms	38	Projects primarily referred to education and outreach programmes that were well-planned before implementation and conducted over a sustained period. They were cited as improving both public opinion and awareness, as well as attracting media attention.
Multi-disciplinary collaboration	32	Projects collaborated and involved organisations with a diversity of expertise, providing a greater breadth of practical and technical skills with which to inform robust scientific planning as well as implementing actions. Collaborators were across sectors, and disciplines and included local, national and international partners.
Community involvement	27	Projects cited the inclusion of local views and representatives in planning as being linked to improved relationships with local communities and often led to increased resources at the project's disposal. Additionally, shared local knowledge gave projects insights into the local area.
Public support	23	Projects found public support increased participation in the project, made it easier to enforce restrictions and provided funding opportunities. However rarely cited, how public support was determined or how it was achieved.
Conflict management	17	Projects sought to address conflict during planning, typically through the use of economic incentives and dialogue with impacted parties. This was found to be effective at reducing conflict risk.
Reasons for failure	**Case studies (n=213)**	**Reason description**
Social conflict	42	Projects cited concerns around released animals interfering with current land-use practices, (damaging crops, predation or the introduction of restrictive designations). Hunting released animals particularly occurred with highly mobile species, which quickly leave release areas. This was a factor in negative public opinions towards both projects and species.
Uncoordinated stakeholders	39	Projects commented on divergent priorities among stakeholder groups, with different organisational priorities rather than project priorities influencing decision-making. This was found to reduce effectiveness and ability to deal with problems.
Opposing views	39	Projects cited deep-held views about the focal species, distrust of the conservation group(s), and changing practices that adversely impact the focal species. This appeared to be an issue even if projects implemented consultations and education and outreach strategies.
Under-resourced engagement	37	Projects used insufficient resources in engagement programmes meaning in many cases education or awareness-raising programmes either didn’t happen or were too short to be effective. This had a direct impact on public support and engagement. Commonly this was due to a lack of resource availability, whether money, people, time or infrastructure.
Political and legal barriers	32	Projects associated this with delays to the project due to difficulty obtaining permits or licences and differing motivations between political and conservation priorities, as well as historical animosity between conservationists and governments.
Communication and awareness	24	Projects referred to both a lack of communication and the need for ongoing communication as a challenge. This manifested as difficulty changing misconceptions of the species or project, and a lack of public awareness

and subsequently public buy-in and support for the project, with public interest waning without regular communication.

Table S1.2. The proportion of conservation translocation case studies that evidenced inclusion of any of the social feasibility guidelines, with respect to the publication of the Guidelines, taxonomic group and IUCN statutory region. *indicates significance

	Projects which included social factors at the Feasibility stage (n)	Projects which did not include social factors at the Feasibility stage (n)	Adjusted residuals
Publication of the Guidelines			
Pre-Guidance	71	113	-2.83*
Post 1998 Guidance	143	146	1.18
Post 2013 Guidance	45	32	2.15*
Taxonomic group			
Vertebrates	220	199	4.55*
Invertebrates	15	31	-2.05*
Plants	24	61	-3.78*
IUCN region			
Meso and South America	27	15	2.32*
North America and the Caribbean	68	52	2.37*
Africa	30	20	1.91
West Europe	54	80	-1.81
East Europe, North and Central Asia	7	14	-1.28
South and East Asia	31	32	0.35
West Asia	8	18	-1.70
Oceania	34	60	-2.32*

Appendix 2: Woodland core area location and data Inputs for the Analytical-Hierarchy decision-making process (Chapter Five)

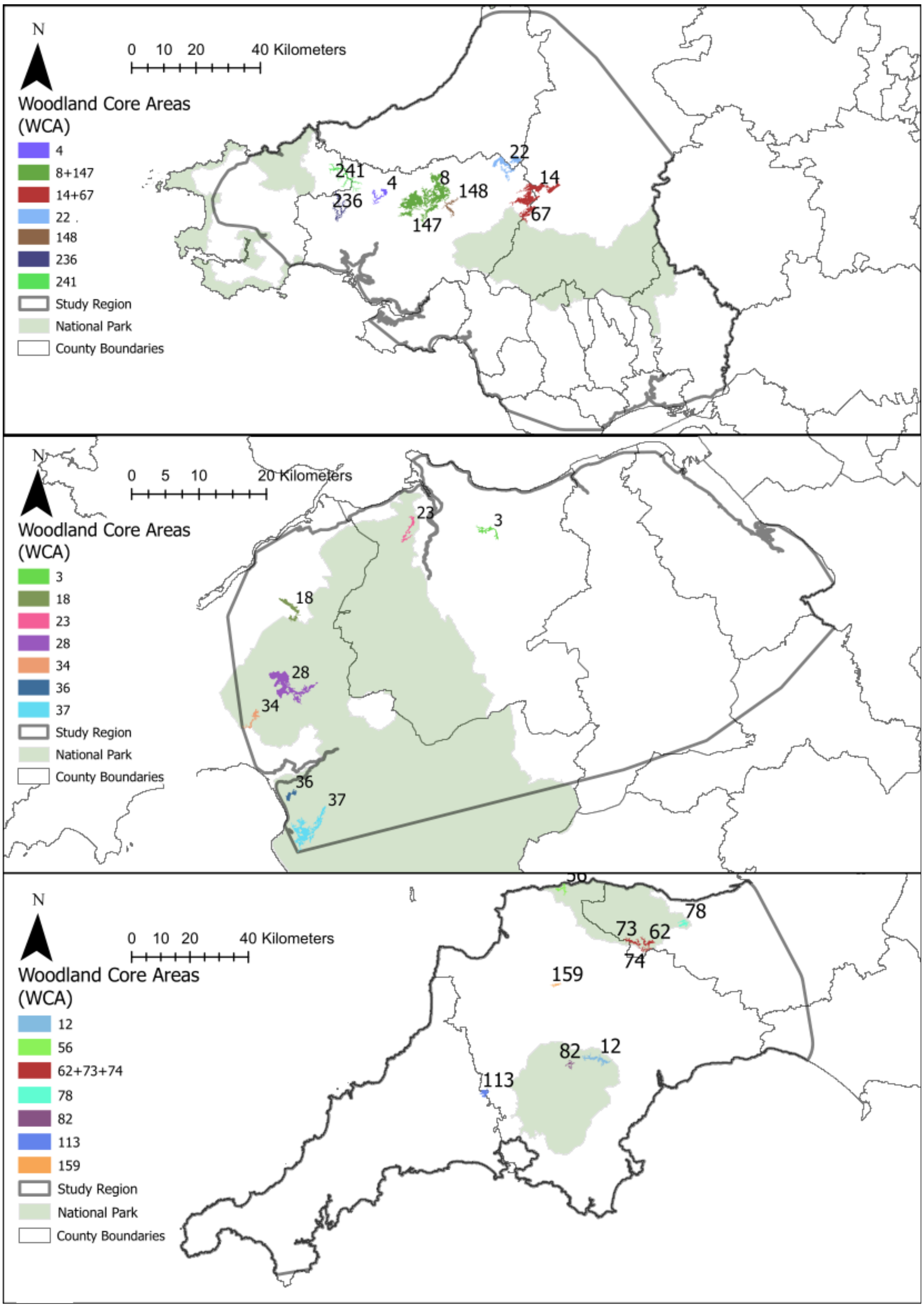

Figure S2.1 The location of woodland core areas (WCA) included as candidate release sites in the Analytical Hierarchy Process (AHP) analysis for West Wales (Top), North Wales (Middle) and South West England (Bottom)

Table S2.1 Raw data for each study region and candidate release site used as inputs for the AHP decision-making.

Region	WCA	Connectivity	Domestic cat density (per/km²)	% suitable habitat	Road density (km/km²)	Game bird and poultry holdings density (per/100km²)	Size (km²)
West Wales	WW8+WW147	8056.38	10.39	39%	0.09	23	72.79
	WW148	5705.07	6.87	35%	0.03	13	4.98
	WW22	2016.87	1.14	24%	0.01	10	17.34
	WW4	5625.84	15.60	41%	0.14	22	6.22
	WW14+WW67	4210.98	3.70	28%	0.12	7	40.57
	WW241	5591.84	14.30	29%	0.09	26	9.77
	WW236	2522.47	14.61	24%	0.00	25	8.33
	Total	**4818.49**	**9.52**	**31%**	**0.07**	**18.00**	**22.86**
North Wales	NW28	195.91	1.30	21%	0.17	6	9.50
	NW3	146.72	45.70	24%	0.13	12	1.50
	NW34	27.13	1.68	9%	0.15	8	1.34
	NW37	26.00	14.70	20%	0.12	3	7.51
	NW23	130.51	75.10	18%	0.11	16	1.54
	NW18	61.90	19.30	19%	0.18	8	2.55
	NW36	26.00	20.17	15%	0.11	4	1.07
	Total	**87.74**	**25.42**	**18%**	**0.14**	**8.14**	**3.57**
South West England	SWE62,73+74	2198.46	11.02	29%	0.10	30	24.42
	SWE159	2408.13	6.87	26%	0.18	32	1.66
	SWE82	1532.72	21.52	31%	0.08	22	2.61
	SWE113	2016.05	23.10	23%	0.06	36	4.63
	SWE12	1077.78	48.60	44%	0.15	32	9.75
	SWE78	1032.64	16.28	24%	0.09	28	3.80
	SWE56	975.99	23.20	20%	0.23	17	4.26
	Total	1605.96	21.51	28%	0.13	28.14	7.30

Appendix 3: Single species occupancy model outputs for each species and site (Chapter Six)

Table S3.1 Occupancy probability with 95% confidence intervals for sites and species and its relationship with covariates *=significant negative relationship, **=significant positive relationship

Site	Species	Detection Probability	Occupancy Probability Ψ	Building distance (m)	Woodland %
England	Cat	0.38 (0.21-0.59)	0.24 (0.19-0.37)	−2.17 (−3.75-−0.60)*	0.76 (−0.56-0.72)
	Fox	0.53 (0.43-0.63)	0.66 (0.55-0.80)	−0.14 (−0.70-0.42)	−0.20 (−0.77-0.35)
	Badger	0.43 (0.27-0.60)	0.27 (0.14-0.39)	−0.35 (−1.01-0.29)	0.29 (−0.32-0.90)
Wales	Cat	0.46 (0.26-0.67)	0.13 (0.06-0.22)	−2.83 (−4.93-−0.72)*	−3.22 (−5.60-−0.75)*
	Fox	0.52 (0.43-0.61)	0.69 (0.56-0.79)	−0.18 (−0.68-0.30)	−1.34 (−2.27-−0.40)*
	Badger	0.48 (0.35-0.62)	0.30 (0.19-0.41)	0.01 (−0.47-0.50)	−0.12 (−0.61-0.36)
	Pine marten	0.20 (0.07-0.45)	0.26 (0.09-0.55)	1.29 (0.43-2.14)**	2.39 (0.10-4.67)**
Scotland	Cat	0.30 (0.21-0.41)	0.25 (0.15-0.37)	−0.30 (−0.83-0.22)	−0.01 (−0.49-0.46)
	Fox	0.29 (0.19-0.46)	0.44 (0.28-0.63)	−0.01 (−0.55-0.51)	0.19 (−0.37-0.75)
	Badger	0.62 (0.51-0.71)	0.30 (0.24-0.40)	−0.56 (−1.17-0.03)	0.00 (−0.47-0.47)
	Pine marten	0.35 (0.27-0.44)	0.62 (0.46-0.74)	1.84 (0.40-3.27)**	0.21 (−0.20-0.63)
	Hybrid	0.35 (0.27-0.44)	0.39 (0.26-0.57)	−0.22 (−0.67-0.21)	−0.01 (−0.45-0.42)
	Wildcat	0.39 (0.31-0.48)	0.47 (0.33-0.59)	0.01 (−0.39-0.41)	−0.25 (−0.69-0.19)

	Species	River distance (m)	Livestock holding density	Elevation (m)	Slope
England	Cat	−0.55 (−1.33-0.23)	0.16 (−0.47-0.80)	−0.71 (−1.53-0.11)	0.52 (−0.14-1.18)
	Fox	−0.10 (−0.65-0.45)	−0.57 (−1.20-0.06)	0.25 (−0.36-0.87)	0.15 (−0.01-0.60)
	Badger	−1.11 (−2.04-−0.17)*	−0.10 (−0.68-0.47)	−0.50 (−1.18-0.16)	0.46 (−0.12-1.04)
Wales	Cat	0.01 (−0.63-0.67)	3.93 (−184.19-192.07)	−0.54 (−1.28-0.19)	−0.90 (−1.75-−0.05)*
	Fox	0.32 (−0.26-0.91)	0.25 (−0.19-0.71)	−0.03 (−0.58-0.51)	−0.44 (−0.99-0.11)
	Badger	0.15 (−0.32-0.62)	0.33 (−0.30-0.98)	−0.07 (−0.55-0.41)	−0.23 (−0.73-0.26)
	Pine marten	−0.63 (−1.58-0.32)	−4.54 (−56.54-47.44)	1.37 (0.53-2.20)**	0.22 (−0.51-0.96)
Scotland	Cat	0.16 (−0.30-0.63)	0.22 (−0.26-0.71)	−0.35 (−0.79-0.17)	0.05 (−0.42-0.53)
	Fox	−0.70 (−1.35-−0.04)*	0.08 (−0.46-0.63)	−0.3491 (−1.00-0.31)	−0.10 (−0.66-0.45)
	Badger	0.04 (−0.43-0.52)	1.22 (0.50-1.93)**	−0.95 (−1.56-−0.31)*	0.01 (−0.46-0.49)
	Pine marten	0.80 (−0.21-1.81)	0.01 (−0.52-0.55)	0.52 (−0.18-1.23)	0.01 (−0.61-0.62)
	Hybrid	−0.24 (−0.68-0.20)	−0.52 (−1.04-0.01)	−0.07 (−0.51-0.36)	0.32 (−0.13-0.79)
	Wildcat	−0.47 (−0.93-−0.01)*	−0.74 (−1.27-−0.21)*	0.24 (−0.18-0.66)	0.12 (−0.29-0.55)

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