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Quantifying and enhancing biodiversity at Irish windfarms

by

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Declaration

I declare that this thesis has not been submitted as an exercise for a degree at this or any other university and it is entirely my own work. I confirm that the work has been completed in full compliance with relevant university policies, including the Academic Integrity policy, the Trinity Policy on Good Research Practice, and the guidance on AI and GenAI in Teaching, Learning, Assessment and Research. I agree to deposit this thesis in the University's open access institutional repository or allow the Library to do so on my behalf, subject to Irish Copyright Legislation and Trinity College Library conditions of use and acknowledgement. I consent to the examiner retaining a copy of the thesis beyond the examining period, should they so wish (EU GDPR May 2018).

A handwritten signature in black ink, appearing to read 'Emma King', is written on a light-colored, textured rectangular background.

Emma King

Summary

As reliance on wind energy is expanding rapidly to address climate change, it is essential that this transition does not place further pressure on biodiversity. Drawing on a transdisciplinary approach, this thesis provides the first comprehensive study on the impact of wind energy on Ireland's habitats and biodiversity, examining barriers and the potential for industry-led mitigation and enhancement measures. In doing so, it advances scientific understanding and practical decision-making around biodiversity at Irish windfarms. In Chapter 2 I review impacts of wind energy on biodiversity in the scientific literature and compared this with how these impacts are addressed in the planning stages of windfarm development. This analysis highlights gaps in our understanding of windfarm impacts across taxonomic groups, and how this reduces opportunities for mitigation in windfarm planning. Using a modified ecosystem accounting framework, I present in Chapter 3 the first site-scale biodiversity-themed ecosystem accounts, demonstrating their utility in identifying areas of high biodiversity importance on windfarms, alongside areas requiring management to further support biodiversity. Building on this, Chapter 4 describes how I combined several datasets with stakeholder engagement to develop evidence-based biodiversity guidelines tailored to the wind energy industry. This work established a robust, scientific foundation to support the implementation of biodiversity mitigation and enhancement at windfarms. To further examine biodiversity guidelines for wind energy, Chapter 5 examines whether bats, a species group thought to be particularly vulnerable to wind energy, utilise areas in close proximity to turbines. Evaluating bat activity across spatiotemporal scales demonstrated the value of context-dependent guidelines and biodiversity monitoring – which I identified as a critical industry knowledge gap. Collectively, the research described in this thesis underscores the importance of utilising a systems approach to understand the biodiversity surrounding Irish windfarms, including acknowledging the gaps in both research and industry practise. Practically, this thesis presents methodologies for decision-support frameworks to guide the Irish wind energy industry in taking meaningful action for biodiversity. Ultimately, this research challenges the common perspective of wind energy and biodiversity as inherently in conflict, demonstrating that, through evidence-based practice and engaged research, windfarms can operate within an iterative, adaptive management framework that advances both climate change mitigation and biodiversity restoration.

In memory of

Aoibheann Gaughran

whose passion for nature & this project motivated me to keep going

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Additional work

Publications:

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- *Wind Energy Ireland Annual Conference*, Dublin (2025) – Oral presentation
- *8th Conference on Wind energy and Wildlife impacts (CWW)*, Montpellier, France (2025) – Oral presentation & panel discussion
- *11th World Conference on Ecological Restoration (Society of Ecological Restoration) Conference*, Denver, USA (2025) – Oral presentation

Chapter 1 General Introduction

1.1 The intertwined crises of climate change and biodiversity loss

The Earth is facing twin crises of climate change and biodiversity loss (IPBES, 2024). Both crises are intrinsically linked through a complex system of feedbacks, with climate supporting organismal niches and distributions and, in turn, biodiversity regulating climate functioning (Gorman et al., 2023). The benefits of this to our own wellbeing and livelihoods is increasingly recognised through a range of identified ecosystem services (Dasgupta, 2021), including provisioning services (e.g., food), regulating services (e.g., water, pollination), and cultural services (e.g., recreation) (Bennett et al., 2009). Energy supply is a key provisioning service that is essential to supporting modern lifestyles (Espézie et al., 2019). The transition to renewable energy sources is critical to meeting emission reduction targets as part of reducing climate change (Potrč et al., 2021). Globally, renewable energy has experienced rapid uptake over the past decade, with a range of technologies now deployed globally (International Energy Agency, 2025). The wind energy sector has been expanding rapidly, achieving the largest sectoral growth compared to other renewable energy technologies in 2021 (European Commission Directorate-General for Energy, 2025). This growth has been in part due to increased recognition of the scale of the climate crisis, improvements in technology, and the large availability of wind as a resource across the globe (Darwish & Al-Dabbagh, 2020). Future growth will require broad land-use changes to accommodate new wind energy facilities and their associated infrastructure.

The need for land-use change to accommodate onshore wind turbines has come into conflict with the need to protect and designate areas for biodiversity conservation, creating a dilemma in which mediating one crisis creates potential conflict with the other. This conflict has been termed the Green-Green Dilemma (Voigt et al., 2024). In spite of growing awareness of this dilemma as global production of wind energy continues to accelerate, our understanding of the impacts of windfarms on biodiversity, particularly at ecosystem- and population-level scales, remains remarkably poor. More needs to be done to understand and improve the ecological

sustainability of onshore renewable energy (Voigt et al., 2019). This necessitates thorough understanding of the impacts on biodiversity to both mitigate those impacts and promote biodiversity-positive design (Kuvlesky et al., 2007).

1.2 Impacts of wind energy on biodiversity

The development of onshore wind energy has been growing since the early 1990s. Soon after the first wind farms were built, what was once believed to be a ‘clean’ source of energy began to be perceived differently as negative environmental impacts began to emerge (Tabassum-Abbasi et al., 2014). In the early 21st century, large numbers of raptorial bird fatalities were discovered at operational windfarms in California in the United States of America (USA) (Thelander et al., 2003) and Spain (Barrios & Rodríguez, 2004). These incidents led to an increase in scientific research into bird collision and their fatality risk at onshore windfarms, with research focusing on identifying potential causes in occurrence (Drewitt & Langston, 2006), estimations of fatality rates (De Lucas et al., 2008), and the introduction of mitigation measures to reduce risk (Smallwood & Bell, 2020). The subsequent focus on monitoring of bird collisions subsequently led to the discovery that bat species were also vulnerable to collision with wind turbine blades (Johnson et al., 2003). Since this discovery, bat collision has been particularly well-documented in the USA (Kunz et al., 2007) and these effects are beginning to be understood in Europe (Gaultier et al., 2020). The discovery and recognition of these impacts have been used as an argument against the deployment of wind energy technology (González et al., 2016).

The discovery of bird and bat fatalities at windfarms has led to increasing demand to further develop our understanding and knowledge of the potential impacts wind energy generation can have on biodiversity (Dai et al., 2015; Nazir et al., 2020). Since the initial discovery of collision fatalities, research has largely focused on understanding why collision occurs and associated mitigation strategies, leading to a large body of research focused specifically on collision of bird and bat species (Arnett et al., 2015; Arnett & May, 2016; Marques et al., 2014; Wang & Wang, 2015). Despite this, there remain many unanswered questions, such as fatality rates (Bernardino et al., 2013), attraction to the turbines as roosting or feeding opportunities (Cryan & Barclay, 2009; Cryan et al., 2014; Pescador et al., 2019), and context-dependent

scenarios such as site-specific confounding effects (e.g., landscape features, blade visibility, turbine configuration, and species abundance) (Barré et al., 2023; Marques et al., 2014; Sotillo et al., 2024). Given the global need to increase reliance on renewable energy, research is shifting to develop methods for mitigating against further losses, building upon the existing understanding of the context of collisions.

While there has been growth in our understanding of impacts on vulnerable bird and bat species in the twenty years of growth of the onshore wind energy industry, it has been hampered by ecological uncertainties in terms of quantifying and understanding population dynamics and population declines (Seifert et al., 2025). An additional complexity arises from generally unstandardised regulatory and policy requirements. This makes comparison between regions difficult, adds complexity when determining whether there are concurrent impacts for migratory species (Conkling et al., 2021; Seifert et al., 2025), and causes difficulties in disentangling the extent to which population-level impacts are directly attributable to wind energy generation (Katzner et al., 2025).

In addition to collision with turbines, there is evidence of additional impacts on bird and bat species. Barotrauma, caused by the rapid decline in air pressure in the proximity of turbine blades has been suggested to cause fatalities, particularly of bats (Baerwald et al., 2008). Field evaluation of this is lacking however, and claims of its occurrence have been debated (Lawson et al., 2020; Rollins et al., 2012). More recently, research has also begun to assess indirect impacts of wind turbines (Pereira et al., 2018) and their associated infrastructure (e.g., roads, hardstands) on other taxonomic groups. Indirect impacts include habitat alteration (Barré et al., 2018; Dohm et al., 2020; Leroux et al., 2024; Skarin et al., 2015), habitat fragmentation (Kuvlesky et al., 2007), trophic alterations (Scholz et al., 2025b; Thaker et al., 2018), and disturbance (Costa et al., 2018; Keehn & Feldman, 2018). Indirect impacts can be restricted to the local scale, such as micro-climate alterations at the turbine level (Armstrong et al., 2016a) but can also affect the entire windfarm and surrounding areas. Such indirect impacts are not unique to windfarms, however, and can also occur through other means of energy generation, such as habitat alteration from oil extraction (Kuvlesky et al., 2007), or changes to vegetation and microclimate from

solar energy (Armstrong, et al., 2016b) requiring shared learning among technologies (Gorman et al., 2023).

It is now widely accepted that windfarms can have impacts throughout all stages of their lifecycle – from construction [habitat alteration, soil compaction (Chen et al., 2019; Xie et al., 2014)], operation [collision, displacement (De Lucas et al., 2008; Tolvanen et al., 2023)], repowering (Huso et al., 2021; Smallwood, 2017), and decommissioning [disturbance, habitat alteration (Hall et al., 2020; Mello et al., 2022)]. Different stages of windfarm lifecycles impact different species and habitats disproportionately. For example, vegetation is heavily impacted during the construction phase (Urziceanu et al., 2024), whereas flying organisms experience greater impacts during operation (Arnett & May, 2016; Voigt, 2021). The end-stages of a windfarm remain the most understudied in terms of potential impacts for biodiversity (Seifert et al., 2025). With a lifespan of 20-25 years (The Welding Institute, 2025), many windfarms are only beginning to reach this stage. It will, therefore, be important to understand how biodiversity and restoration planning can become an integrated part of this life stage such that the land is restored to its original biodiversity and ecosystem functioning.

Cumulative impacts and an ecosystem-focused approach to understanding windfarm impacts on biodiversity remain understudied in the scientific literature. There are some emerging case-studies that document impacts of windfarms on trophic interactions (Thaker et al., 2018), community composition (Karamanlidis & de Gabriel Hernando, 2025), and the introduction of invasive species (Keehn & Feldman, 2018). Little research has, however, been done on how the installation and operation of windfarms can affect surrounding areas or on how multiple windfarms in a landscape interact to impact populations (Dhunhy et al., 2019). Moreover, potential benefits of ecosystem restoration alongside renewable energy generation are largely unknown (Seifert et al., 2025).

Upon the onset of the discovery of large-scale bird and bat collisions, research began to focus on mitigation. Most mitigation measures focus on collision avoidance, with best-practice measures that are applicable in windfarms across jurisdictions including avoidance of sensitive areas, repowering with larger turbines, grouping

turbines to provide corridors, curtailment, acoustic deterrents, and compensatory measures, which are measures taken typically to replace habitat that is lost during construction or operation when damage to said habitat is unavoidable (Arnett & May, 2016; Drewitt & Langston, 2006). Incorrect siting is the primary driver of collision risk (Voigt et al., 2024). Consequently, correct planning and siting at the preconstruction phase can provide particularly beneficial mitigation against collisions (Arnett & May, 2016) and has received much research focus in recent years. Improved technology for remote sensing has contributed to substantial advances in species distribution modelling that can help developers identify suitable locations that avoid high traffic areas for vulnerable species (Froidevaux et al., 2025; Santos et al., 2013; Wimhurst et al., 2023).

Curtailment is the modification of turbine operation or schedule to minimise the risk of collisions for birds and/or bats. There are two curtailment types. Blanket curtailment involves shutting down turbines under specified weather conditions or time periods (e.g., migration periods), whereas smart curtailment utilises technology to make near real-time decisions regarding turbine shutdowns (Smallwood & Bell, 2020; Tomé et al., 2017). Curtailment can also involve raising the cut-in speed, which is defined as the lowest wind speed at which turbines rotate to generate power, which essentially reduces turbine operation at low wind speeds (Arnett et al., 2011; Baerwald et al., 2009). This approach has shown to be successful for bats but not birds (Smallwood & Bell, 2020). Comparison of this approach across multiple curtailment experiments showed that bat fatalities could be reduced by 62% by increasing the cut-in speed (Whitby et al., 2024). More recently, there has been a shift towards smart curtailment or shut-down on demand approaches, which utilise detection systems and technology to make near real-time curtailment decisions at the turbine level, benefitting both vulnerable species and energy production (Hayes et al., 2019; Tomé et al., 2017). Smart curtailment utilises many different approaches, including acoustics (Hayes et al., 2019; Peterson et al., 2025), model-based approaches (Behr et al., 2017), and radar detection systems (Tomé et al., 2017). The estimated effectiveness of curtailment is only as good as the ability to detect fatalities. More recently, collision detection dogs have been utilised, with much higher success rates

than human searchers (Domínguez del Valle et al., 2020; Paula et al., 2011; Smallwood et al., 2020). Advancements in technology, such as the use of radar (Aschwanden et al., 2018) and 3D thermal video (Matzner et al., 2020), have also improved fatality monitoring. Such tools improve understanding of the success (or failure) of mitigation measures compared to previous human-based searches and extend the temporal and spatial capacity of searches substantially. However, as the vast majority of research on mitigation has been on bird and bat collisions, there is a pressing need for research on mitigation opportunities for other taxonomic groups (Kuvlesky et al., 2007). Rapid technological advancements over the past decade highlight how technology can advance understanding and applying these approaches to other taxonomic groups could highlight restoration opportunities.

1.3 Wind energy in Ireland

The wind energy sector in Ireland has been growing rapidly and continuously over the past two decades, achieving a mean annual growth of 25.9% in that time (SEAI, 2024). In 2024, 32% of Ireland's electricity came from wind across 300 onshore windfarms in the Republic of Ireland (Wind Energy Ireland, 2024). Onshore windfarms are located within various land use types, including agriculture, forestry, and peatland. Over time, the number of windfarms has been increasing in agricultural settings and declining in peatland habitats. There has also been a trend towards larger turbine heights, coupled with fewer turbines per windfarm (Hallan & González, 2020). The growth in the wind industry in Ireland is supported by national policy contexts, which sets renewable energy as a key policy priority, with a commitment to achieving 80% of Ireland's electricity generation from renewable sources by 2030 (Department of the Environment, Climate and Communications, 2024). This goal is important in meeting Ireland's climate change commitments, which are inextricably linked with its biodiversity commitments, as in 2019 Ireland declared joint climate change and biodiversity emergencies (Oireachtas, 2019).

While current policies are supportive of renewable energy and its development in Ireland, support for biodiversity restoration and protection during this process is inadequate, and new European Union legislation such as the Nature Restoration Law may bring potential conflict with the need to increase reliance on renewable energy

(Jackson, 2011; Rehbein et al., 2020). Further challenges facing the wind energy industry in relation to biodiversity includes the lack of focused guidelines on the topic from a national level, with the current Wind Energy Development Guidelines in force dating from 2006 (Department of Housing, Local Government and Heritage, 2006). Attempts to update this set of guidance in 2019 were not introduced and are still undergoing significant revisions which have not been finalised, despite the issue being raised on multiple occasions in Dáil Éireann (Oireachtas, 2025b, 2025d, 2025d). Lack of up-to-date guidance has led to uncertainty over biodiversity enhancement within the industry. Combined with the rapid expansion of the industry, this has led to mixed interpretation and industry members using available guidance from other areas (e.g., Scotland's NatureScot's guidance documents (NatureScot, 2023), reducing motivation for industry members to implement initiatives for biodiversity (Seifert et al., 2025).

The lack of industry-specific biodiversity policy undermines the potential for the wind energy industry to contribute to biodiversity-related initiatives. Consequently, the primary mechanism for protecting biodiversity surrounding wind energy in Ireland is during the planning stage within Environmental Impact Assessment (EIA). This is a planning process that combines project descriptions and environmental baselines to predict potential impacts of a development (Glasson et al., 2012). All European Union (EU) Member States require projects to produce an Environmental Impact Assessment Report (EIAR) when a development is likely to have a significant effect on the environment due to its size and/or nature (Council Directive 2011/92/EU, 2011). For onshore windfarms, an EIA is required when a proposed project is over 5 megawatts capacity or consists of five or more turbines (Section 172, Planning and Development Act, 2000). Within the EIA, the potential impacts on many environmental components must be assessed, including biodiversity, geology, water and cultural heritage. The assessment process for onshore windfarms in Ireland typically involves habitat mapping, targeted surveys for bird species and bat species if the surrounding habitat indicates they could be present, and for protected species that may be present in the area. The aim of the EIA is to avoid impacts prior to any development beginning. Where avoidance is not possible, the subsequent steps of the mitigation hierarchy, which is a

priority framework for minimising negative environmental impact, are followed: minimise, restore and offset. Where the EIA process fails to correctly predict biodiversity impacts, which is not uncommon (Briggs & Hudson, 2013; Lintott et al., 2016), opportunities for monitoring and adaptation for biodiversity mitigation and protection in the post-planning stages are highly constrained. Moreover, the EIA process provides minimal opportunity for biodiversity enhancement throughout the lifespan of windfarms.

There has been little research focus on how Irish biodiversity has been impacted by onshore wind energy. Since Ireland's first windfarm was built in the 1990's, there have been few published studies on Irish biodiversity's interaction with wind energy. Those studies that have been done focused primarily on birds (Fernández-Bellon et al., 2019; Wilson et al., 2017) and impacts on peatland (Chico et al., 2023). Consequently, much of the current understanding to prepare for EIAs utilises knowledge from the United Kingdom. The main impacts that have been identified include collision risk for birds of prey (Fernández-Bellon et al., 2019). For Irish bats, most species are regarded to be of relatively low collision risk (Newson et al., 2017), due to their generally small size and ecology. There has been little research into other impacts, such as habitat loss or displacement. Similarly, plants and invertebrates are typically included within the EIA process only when a species is protected. Inappropriate siting of windfarms on peat habitats in Ireland has triggered several peat landslides, such as the Derrybrien Bog slide in Galway (Lindsay & Bragg, 2005). Such events cause serious long-term damage to the structure and function of the peat, which negatively impacts bog vegetation and the surrounding environment. These events harm social perception of wind energy, creating substantial community opposition towards new developments (González et al., 2016).

While knowledge gaps can be inferred from other countries with similar habitats, potential impacts nonetheless need to be mitigated and managed with limited data and understanding. This has led to the recommendation that Ireland follow an adaptive management approach to improve biodiversity on onshore windfarms (Köppel et al., 2014). Adaptive management is an iterative approach to management and decision-making based on continuous learning and consideration

of the consequences of the actions (Holling, 1978; Walters & Hilborn, 1978). An adaptive management approach requires transparency, consistent monitoring, and a willingness to adapt (Williams, 2011). Adaptive management can also benefit from a transdisciplinary approach between industry members and ecologists. Use of an adaptive management approach for mitigation and enhancement of biodiversity will improve the knowledge base, allowing for improved iterations of actions over time. Prior to implementing adaptive management there needs to be an understanding of potential impacts. In Ireland, there is a large research gap on specific impacts on Irish habitats and species, much of our current understanding comes from the planning stage, specifically the EIA process. These reports provide a huge volume of data, and to our knowledge there has been no review of the biodiversity chapters of Irish windfarm EIARs to assess these data and potential impacts that are accounted for. Currently, even for a protected species group with well-documented impacts globally from wind turbines, there is little information regarding Irish species collision risks. Due to the available data being restricted to the planning stage, there is a large gap in understanding how habitats surrounding windfarms change post-construction and how biodiversity surrounding operational windfarms is managed or impacted by the operations of the windfarm. This means data are limited when choosing which actions to take when choosing how to mitigate or enhance local biodiversity. There is, therefore, a need for standardised, spatially-explicit tools that can measure local ecosystems, and can be scaled to various temporal, spatial scales, and include different ecosystem services. Following from this, industry members may not always have available ecological expertise to advise on what mitigation or positive actions to take, and choosing the incorrect action even with the best intentions can have negative consequences for biodiversity. Therefore, decision-support tools are needed to aid industry in choosing the correct options and ultimately should support the uptake of positive action and reinforce an adaptive management approach.

1.4 Thesis outline

The motivation for this thesis is to understand how biodiversity protection and renewable energy expansion can coexist in Ireland to jointly address the intertwined crises of climate change and biodiversity loss. The chapters of this thesis are

structured around four interconnected elements that advance both scientific understanding and—critically—practical decision-making for biodiversity at windfarms: understanding impacts (Chapter 2), measuring biodiversity (Chapter 3), managing biodiversity (Chapter 4), and monitoring vulnerable species (Chapter 5).

In Chapter 2, I explore how wind energy impacts nature in terms of how it has been studied within the scientific literature and compare this to how different taxonomic groups are represented within wind energy planning in Ireland. I found that over time taxonomic diversity has increased in both scientific literature and EIARs, however understudied taxa remain at risk of being under-surveyed within EIARs, and there needs to be greater knowledge sharing of biodiversity impacts to increase overall understanding.

In Chapter 3, I describe a framework for ecosystem accounting at the scale of individual windfarms, specifically focused on the theme of biodiversity, as a tool for understanding the surrounding habitats and their condition on windfarms. Ecosystem accounting highlighted the biodiversity potential of some habitats, particularly peatland, marsh, and freshwater systems and the spatially-explicit methodology identified locations where improved management could benefit biodiversity.

In Chapter 4, I describe the process of developing site-specific biodiversity action plans and then scale these into a set of sectoral guidelines to protect and enhance biodiversity at windfarms in Ireland, with the aim of this becoming a resource used by industry. I assess and prioritise biodiversity actions using stakeholder engagement and a cost-effectiveness analysis. This process highlighted the importance of combining both scientific and stakeholder engagement to develop decision-support tools for industry.

In Chapter 5, I investigate bat activity at onshore windfarms in Ireland using acoustic data. I explore whether bats are feeding in close proximity to turbines and the influence of weather and habitat variables on their activity rates and feeding behaviour. Monitoring bats at operational windfarms revealed scale-dependent effects of drivers on bat activity and species richness, underscoring the need for context-dependent guidance regarding siting and management of habitats around wind turbines.

In Chapter 6, I summarise and discuss my overall findings, focusing in particular on the ability of windfarms to contribute towards biodiversity restoration and enhancement, the motivations for the industry to do so, and how we can combine transdisciplinary approaches to provide tools for the industry to support decision-making.

Chapter 2 Biodiversity biases in research and practice: lessons from the Irish onshore wind sector.

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Author contribution: EK and ID conceived the ideas and designed the methodology; EK collected and analysed the data; EK and ID led the writing of the manuscript. CG, JCS and YMB assisted with drafting the manuscript. All authors contributed critically to the drafts and gave final approval for publication.

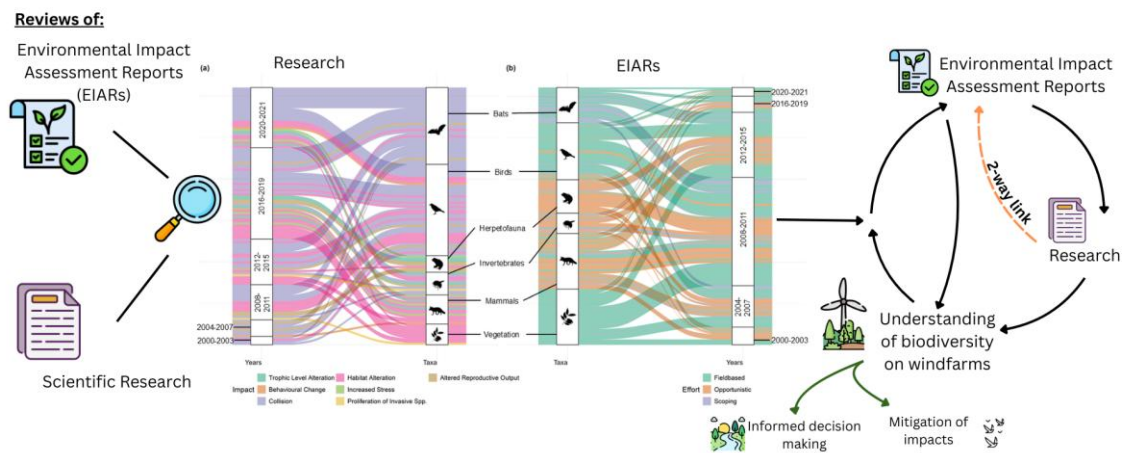
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2.1 Abstract

Alongside increases in renewable energy developments associated with climate change mitigation efforts, there has been increasing recognition of the role of biodiversity in mitigating and adapting to climate change. Environmental Impact Assessment Reports (EIARs) document potential impacts of proposed new developments on a broad range of environmental factors, including biodiversity, and propose actions to alleviate those impacts. More recently, addressing trade-offs between developments of infrastructure for climate mitigation and biodiversity impacts in EIARs is becoming increasingly important. However, biodiversity may be measured in many ways, impacts may be taxon-specific, and there can be survey biases towards particular species and taxonomic groups. It is, therefore, important to consider what is included in EIARs, and to understand the extent to which the taxonomic focus of EIARs aligns with published scientific research evidence. Here, we systematically review both ecological surveys conducted in EIARs of granted windfarm applications in Ireland between 2000 and 2021 and the scientific literature examining the impacts of onshore windfarms on biodiversity. We found that EIARs in the early 2000's examined a considerably more diverse range of animal and plant groups than the scientific literature at that time. This divergence in focus diminished through time as both EIARs and the scientific literature captured a more diverse range of taxa. However, taxa and impacts with low prominence in the scientific literature were also surveyed less frequently in EIARs, highlighting that understudied taxa and biodiversity

impacts are at risk of being underestimated or undetected at the development stage. We conclude that explicit comparison of the two-way link between scientific literature and EIARs can aid in identifying knowledge gaps and assessment of the broader impacts of renewable energy developments, helping both to inform appropriate mitigation for biodiversity impacts and to inform policymakers.

2.1.2 Graphical Abstract



2.2 Introduction

Globally, there is a critical need to substantially reduce emissions of greenhouse gases to mitigate the impacts of unimpeded climate change (IPCC, 2023). At the same time, we are faced with an accelerating biodiversity crisis and need equally urgent action to prevent further biodiversity loss (IPBES, 2016). Accordingly, there is increasing appreciation that we must avoid putting further pressure on biodiversity as we act to mitigate climate change (Gorman et al., 2023). The transition to renewable energy sources is critical to meeting emission reduction targets to mitigate climate change (Potrč et al., 2021). Though renewable energy installations are known to have negative consequences for some species (Rehbein et al., 2020), some of these impacts can be effectively mitigated throughout the development, operation and decommissioning stages (Gorman et al., 2023). Mitigation measures should be identified and planned for in the early stages of a new development. Planning

documents are often the first assessment of potential impact of a proposed development. As such, they play a significant role in decision-making and can also inform post-planning management actions (Glasson et al., 2012).

Wind energy generated from onshore turbines is one of the most widely used sources of renewable energy around the globe (Gielen et al., 2019). Globally, the wind energy sector has been expanding rapidly (International Energy Agency, 2022) requiring land-use change to accommodate new renewable infrastructure. The development of windfarms can, however, have negative (e.g., Arnett et al., 2015; Helldin et al., 2012) as well as positive (e.g., Pustkowiak et al., 2018; Xu et al., 2019) consequences for terrestrial biodiversity, if appropriate management and mitigation measures are in place. Wind turbines themselves can affect above- (Arnett & May, 2016; Schuster et al., 2015) and belowground biodiversity (Łopucki et al., 2018; Velilla et al., 2021). Associated infrastructure, such as roads and substations, contribute to the overall impacts of wind energy generation. The biodiversity impacts of windfarms vary throughout the stages in their lifespans (that is, construction, operation, and decommissioning). For example, research has shown that habitat displacement can occur during windfarm construction. This can be short-term for some species (Costa et al., 2018) but, for others, such impacts can last throughout the operational phase as a consequence of, for example, operational noise (Klich et al., 2020; Łopucki et al., 2017). Habitat alterations surrounding turbines can also have positive impacts on biodiversity, introducing heterogeneity and resulting in increased pollinator populations (Pustkowiak et al., 2018). During the operational phase, collisions of volant species with turbines, particularly birds and bats, has been highlighted at windfarms across the globe (Arnett & May, 2016), with negative consequences for populations (Duriez et al., 2023) and ecosystems (Thaker et al., 2018). In addition to direct impacts on biodiversity, windfarms can have broader environmental impacts through modifying, for example, the microclimate (Armstrong, et al., 2016a), water quality (Heal et al., 2020) and peatland stability (Lindsay & Bragg, 2005), with knock-on consequences for biodiversity.

Effective mitigation of the negative effects of wind farms on biodiversity requires a comprehensive understanding of potential impacts. Appropriate action at

the development and planning stages provides the best opportunity to avoid many potential negative impacts (Dai et al., 2015). For example, early windfarms in the United States of America were subject to large numbers of bird collisions post-construction due to poor siting and a lack of pre-construction impact assessment data (Smallwood & Thelander, 2008). This led subsequent windfarm developments to include bird surveys in the siting stages of planning (Wang & Wang, 2015). Later, bats were also included in surveys due to the identification of collision impacts during operational surveys for birds at windfarms (Kuvlesky et al., 2007). Applying knowledge gained during planning and operational surveys can, therefore, be critically important for protecting biodiversity. Avoidance is best practice for reducing impacts of renewable energy on biodiversity. Knowing what species are present on sites and how they might be impacted is essential (Arnett & May, 2016).

Environmental Impact Assessment (EIA) is a global systematic process that utilises project descriptions and environmental baselines to predict and identify potential impacts of developments, while also encouraging public consultation (Glasson et al., 2012). EIAs are a tool to aid management decisions for a wide range of developments and they are also used to develop environmental management plans post-planning in some countries (Glasson et al., 2012). All European Union (EU) Member States are required to produce an Environmental Impact Assessment Report (EIAR) when a construction project or new development is likely to have a significant effect on the environment due to its size and nature (Council Directive 92/43/EEC, 1992). In the case of onshore windfarms, an EIAR is required when a development is over 5 MW capacity or consists of five or more turbines [Section 172 Planning and Development Act, 2000 (Government of Ireland)]. In some circumstances, where deemed to have potential impact on the environment, a windfarm planning application that is below the above thresholds may still require an EIAR at the request of the planning authority [Articles 103 120 Planning and Development Regulations, 2001 (S.I. No. 600/2001)]. EIARs must assess the potential for impacts on biodiversity as well as several other components of the environment, such as land, soil, water, and cultural heritage. There has, however, been significant criticism surrounding the EIA process,

in terms of its success at both predicting and mitigating impacts (Jay et al., 2007), and the extent of its inclusion in the decision-making process (Loomis & Dziedzic, 2018).

European Union legislation, together with findings of scientific research, inform the biological and environmental surveys that are included in EIARs to identify potential impacts on biodiversity at a proposed development (Drayson et al., 2017). The EIA Directive (2014/52/EU) references that biodiversity should be incorporated into the assessment, with specific reference that there should be a focus on species and habitats included in the Habitats Directive (Directive 92/43/EEC) and Birds Directive (Directive 79/409/EEC). Reviews of EIARs have found that many species may be omitted from EIA surveys (Drayson et al., 2017) and, moreover, that the magnitude of impacts are often underestimated (Briggs & Hudson, 2013; Lintott et al., 2016b). The presence of undetected or unconsidered species and/or impacts at a site can lead to a lack of mitigation for these species and impacts, contributing to biodiversity decline (Davy et al., 2021). Evidence of impacts and — critically — scientific understanding of mechanisms of impact are, therefore, very important to ensure EIARs appropriately identify the potential for impact at early stages so that appropriate mitigation can be put in place.

Here, we investigate the relationship between scientific research on biological impacts of windfarms and biodiversity assessments in EIARs, with a view to identifying the extent to which taxa included in EIARs correspond to those identified as impacted in published research, using the wind energy sector as a case study. We expect that, if taxa and/or impacts were understudied in the scientific literature, and not protected under legislation, they would also receive poorer survey effort during windfarm development. To address this, we first conducted a review of scientific articles that examined the impacts of onshore windfarms on biodiversity across six focal organism groups (that is, bats, birds, herpetofauna, invertebrates, mammals, and vegetation). We then conducted a systematic search of available EIARs in Ireland over a > 20-year period and assessed variation in the linkages between both sets of data. We focused on the Republic of Ireland as it is one of the leading countries in generating capabilities for onshore wind energy in Europe (SEAI, 2023).

2.3 Materials and Methods

We surveyed the global scientific literature by systematically searching for papers that examined the impacts of onshore windfarms on biodiversity from 2000-2021 (inclusive), using Clarivate's Web of Science database and Google Scholar. We used the search terms "biodiversity impacts", "wind farm(s)", "wind energy", "onshore wind farms", and used the following categories to group focal organisms: bats; birds; herpetofauna; invertebrates (categorised separately into aboveground, belowground and volant); mammals; and vegetation. These categories were based on the typical headings used in Irish EIARs. A full list of search terms is provided in Table 2.1. We also recorded the primary cause of impact in each study (e.g., direct collision, vibration, etc.), where reported.

Table 2.1. Boolean operators and search terms used in our systematic literature review on biodiversity impacts and wind energy. The Boolean operator "OR" was used between synonymous search terms for wind energy. The Boolean operator "AND" was used between these search terms and taxonomic search terms. Taxonomic search terms are organised by our focal taxonomic groups.

Boolean Operator	Search Term (windfarms)	Boolean Operator	Taxonomic Group	Search Term
OR	Wind farm	AND	Bats	Bat
	Wind farms			Bats
	Windfarm			Chiropteran
	Windfarms		Birds	Bird
	Wind energy			Birds
	Onshore wind farm			Avifauna
	Onshore windfarms		Herpetofauna	Herpetofauna
	Onshore wind energy			Reptiles
				Amphibians
			Invertebrates	Invertebrate
				Invertebrates
				Insect
			Insects	
	Mammals		Mammal	
Mammals				

In tandem, we conducted a systematic search of available windfarm EIARs in Ireland. Planning applications are submitted on a local (county-scale) basis. There are over 300 windfarms in the Republic of Ireland (Wind Energy Ireland, 2023), thus providing an ample and searchable dataset. Biodiversity or ecology chapters of EIARs are a standard EU planning document under the Environmental Impact Assessment Directive (Directive 2011/92/EU, as amended by Directive 2014/52/EU). The Directive requires that projects deemed likely to have significant effects on the environment produce an assessment to identify and assess direct and indirect significant effects of a proposed development on biodiversity, with particular attention to the species and habitats protected under the Habitats (Directive 92/43/EEC) and Birds Directives (Directive 79/409/EEC). As part of the planning application, these documents are publicly accessible and provide an opportunity to understand potential impacts that may be prioritised in early development. Mitigation measures and other protection measures may be put in place by operators following operational surveys. However, survey data collected during the operational phase are at the discretion of individual windfarms and are often not publicly available, hence we focused our data collection efforts on pre-planning ecological EIAR chapters.

We collected EIARs on windfarm planning applications from every county in Ireland, using the terms “wind farm”, “windfarm”, and “wind turbine” in searches of <https://eplanning.ie>, a platform that facilitates the search of Irish planning documents through relevant county council websites. We restricted our results to onshore windfarms where planning permission had been granted (including both conditional and unconditional permission) from 2000-2021. For each EIAR, we reviewed the chapters pertaining to biodiversity and/or ecological survey information and collected data on the taxonomic groups that were surveyed and the survey methodology. Survey methodology was grouped into one of four categories (see Table 2.2 for category descriptions).

Table 2.2. Definitions of the survey methodology used in ecological EIAR chapters to survey across six focal taxonomic groups (that is, bats, birds, herpetofauna, invertebrates, mammals, and vegetation).

Nature of Survey	Definition	Example(s)
<i>None</i>	No surveys conducted for the focal taxonomic group/no mention in EIAR.	–
<i>Scoping</i>	Taxonomic group mentioned, information on species presence was attained using remotely obtained data.	Public databases were searched for records and/or for determining likely habitat suitability from remote imagery and understanding of species ecology.
<i>Opportunistic</i>	No planned or focused surveys conducted but the presence or evidence of presence of taxa was noted during site visits.	Opportunistic recording of presence of tracks/signs of focal group(s) when on site during targeted surveys of other groups.
<i>Field-based</i>	Specific, targeted surveys conducted for the focal taxonomic group.	Vantage point surveys for birds of prey. Acoustic recorders placed on site to identify bat presence.

Data Analyses

We quantified the diversity of the taxonomic groups reported in the literature and EIARs using Shannon’s Diversity Index (Shannon, 1948). We used a linear model with an interaction term (year x source) to test whether diversity varied over time between the two data sources. Our linear model was based on the null hypothesis that the diversity of the taxonomic groups was the same between the two sources over time. The model was fit using the `lm()` function in R version 4.2.2 (R Core Team, 2022). Model assumptions were checked using residual diagnostics. Data were visualised

using the ‘ggplot’ (Wickham, 2016) and ‘ggrepel’ (Slowikowski, 2023) packages in R version 4.2.2.

2.4 Results

Our global literature search identified 124 scientific articles relating to impacts of onshore windfarms on biota from 2000-2021 (Table A.1). There were no geographic search restrictions, and published literature was found across five continents (Fig. 2.1a), with most from Europe (48%, 59/124 papers; Fig. 2.1b). Eighteen papers (15%) incorporated results from multiple countries. A full list of studies and their geographic locations are provided in Table A.1.

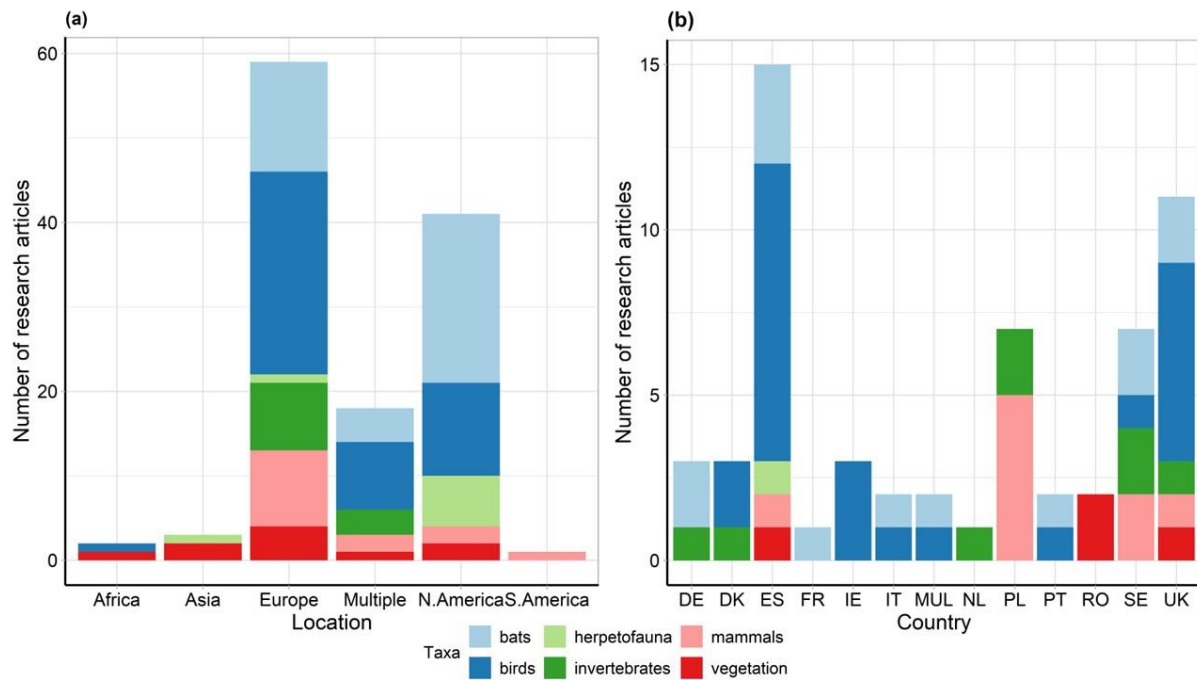


Figure 2.1. Geographical spread of published scientific research relating to impacts of onshore windfarms on biota from 2000 to 2021 across continents (a) and countries in Europe (b). Multiple (MUL) refers to research that includes results across multiple countries or regions: DE—Germany; DK—Denmark; ES—Spain; FR—France; IE—Ireland; IT—Italy; NL—Netherlands; PL—Poland; PT—Portugal; RO—Romania; SE—Sweden; and UK—United Kingdom. We highlight results for Europe as this is the region in which our case study on EIARs is from. Ireland had only three scientific research publications identified in our survey relating to onshore wind impacts on biodiversity, all of which focused on birds.

Within the scientific research identified in our survey, seven general categories of impact were described across our focal taxonomic groups: collision with wind turbines; habitat alteration and loss; proliferation of invasive species; increased stress

(via, for example, increased levels of stress hormone cortisol, or corticosterone); reduced reproductive output; alterations to food webs (e.g., species impacted due to a change in food web or a cascading impact across the food web); and behavioural changes. Early studies (2000-2007) of potential windfarm impacts accounted for just under 10% of the total number of papers in our survey, with studies focusing primarily on collision impacts on birds and invertebrates, and behavioural change in mammals. Collision was the most recorded type of impact in our review overall (52% of studies, 64/124 papers) and research grew notably in this area from 2008 onwards (Fig. 2.2a). This is reflected in a surge in the literature on impacts on volant taxonomic groups, such as birds and bats, from 2008. Habitat alteration was also a common impact category studied in the literature from 2008 onwards (32% of studies, 36/112 papers from 2008-2021). This focused largely on birds, but there were also some studies on mammals and vegetation. More recently, a wider variety of impacts were studied across a greater range of taxa (Fig. 2.2a). From 2016, all six of our focal taxonomic groups were consistently represented in the literature. Moreover, these studies also incorporated a broader diversity of impacts. Though studies of collision impacts remained a priority in studies from 2016-2021 (49% of studies, 36/73 papers), focusing primarily on bats (56% of studies of collision impacts, 20/36 papers), research into all five of the other categories of impact together made up the remainder of these more recent studies (51%, 37/73 papers).

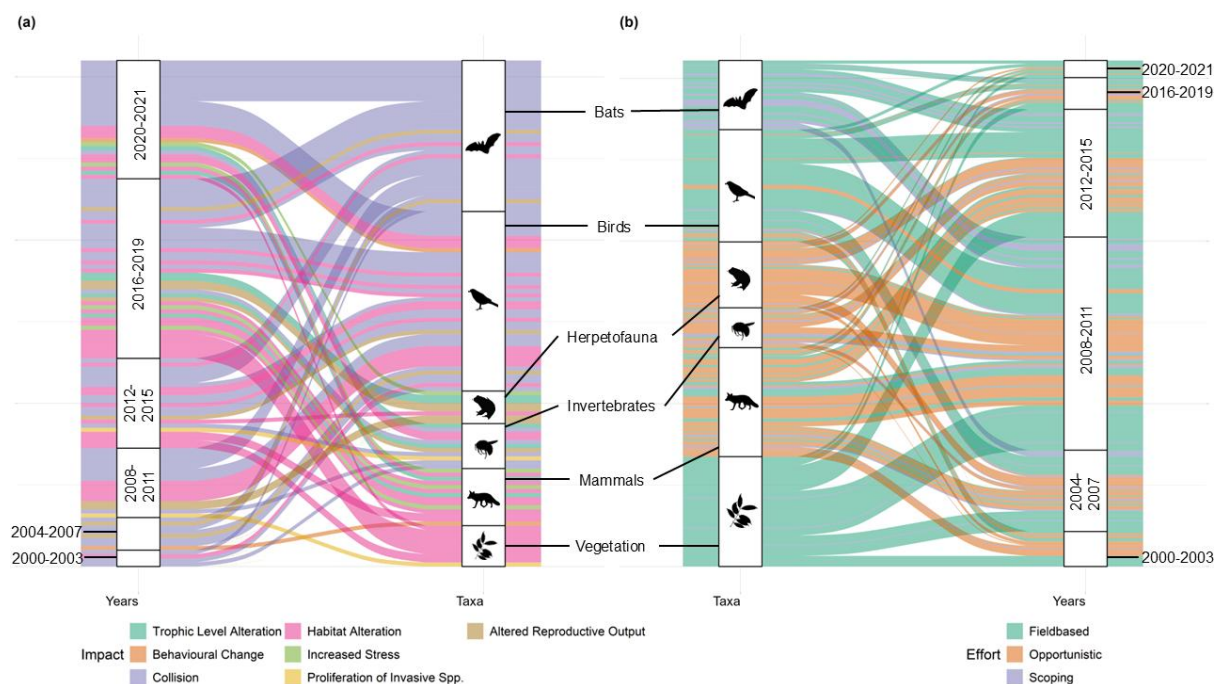


Figure 2.2. Variation over time (2000–2021) in studied taxa and impacts of onshore windfarms on biodiversity from the global scientific literature (a) and the taxa targeted in Irish EIARs through different survey methods (b). Segments on the vertical axes represent categories (years and taxonomic groups) and are proportional in size to the number of published studies (n = 124) or surveys (n = 622). In both plots, the number of research articles and EIARs has increased through time. In the scientific literature (a) the number of research articles focusing on birds and bats is greater than that of other taxonomic groups, while in EIARs (b), the number of surveys is more consistent across the taxonomic groups.

Our survey identified 336 granted windfarm applications, of which 138 (41%) provided an EIAR with an uploaded and accessible ecology chapter. A total of 622 surveys were reported in those applications, with substantial variation among them in the taxa prioritised (Fig. 2.2b). The reasoning for surveying vegetation provided in most EIARs was to provide context on the landscape of the proposed development, in the absence of an Irish national land cover map prior to 2023. Field-based surveys were the most common methodologies used (57% of reports, 354/622 surveys), followed by opportunistic (33%, 205/622 surveys) and scoping (10%, 63/622 surveys) methods. Early EIARs had an uneven distribution of survey methods across taxa but this distribution is changing and becoming more even over time. Recent EIARs are implementing field-based surveys more widely and equitably across groups, with field-based methods accounting for 47% (20/43) of surveys from 2000-2003 but 81% (17/21 surveys) by 2020-2021.

Birds were surveyed in all EIAR chapters reviewed predominantly using field-based methodologies (86%, 119/138 surveys). Field-based surveys were dominated by a focus on vegetation and birds with occasional inclusion of other groups until 2010, when all six of our focal taxonomic groups started to become more represented in EIARs. Herpetofauna were the taxonomic group surveyed most using opportunistic methods (88%, 71/88 herpetofauna surveys), in addition to mammals (65%, 87/134 mammal surveys). There were 49 surveys of invertebrates and the majority of these were also done using opportunistic methods (67%, 33/49 invertebrate surveys). Although invertebrates were originally further categorised into volant, aboveground, and belowground groups, there were insufficient studies and papers to analyse each group separately, and some surveys did not specify which group was included. Scoping surveys were the most inconsistent methodology used in EIARs, and from 2017 onwards no scoping surveys were reported. Bats were the taxonomic group surveyed most frequently using scoping methodologies in Irish EIARs (30/55 bat surveys).

The taxonomic focus of both the scientific literature and EIARs has changed considerably over time (Fig. 2.3a, b). From 2000, the taxonomic coverage of both scientific literature and environmental impact assessment has increased notably, with consistent and more evenly-distributed taxonomic coverage in EIARs relative to scientific research, while the latter also shows notable year-to-year variation in focus (Fig. 2.3). The taxonomic diversity included in both the scientific literature and in EIARs has increased significantly over time ($t = 3.4$, $P = 0.001$; Table A.2). However, the rate of this increase differs between these data sources (interaction between Data Source and Year: $t = 4.47$, $P < 0.001$; Table A.2; Fig. A.1), with taxonomic diversity increasing more steeply over time in the scientific literature compared to the EIARs, though from a lower baseline (Fig. 2.3c).

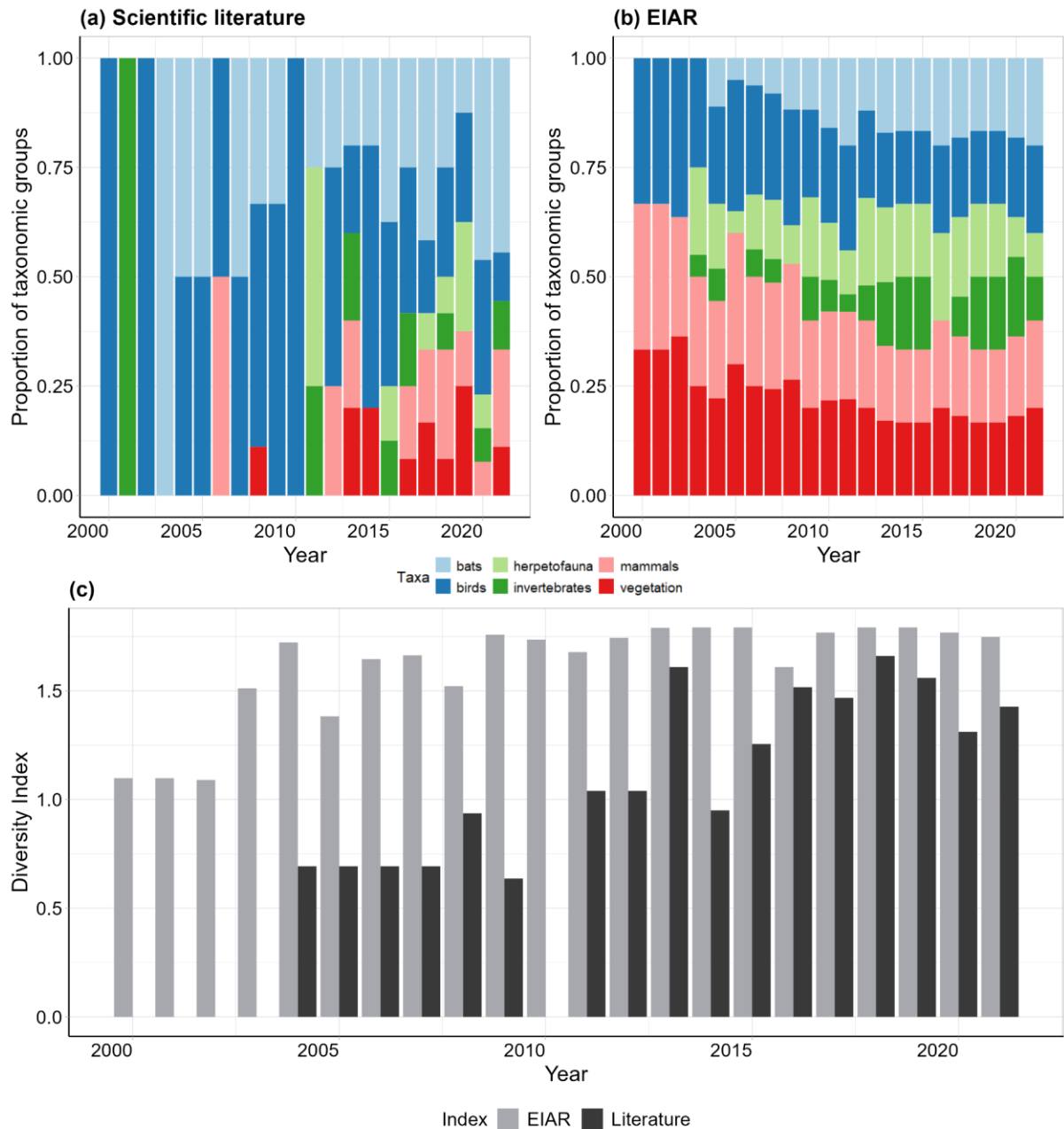


Figure 2.3. Relative taxonomic focus of scientific literature (a) and EIARs (b) over time (2000–2021), and the (Shannon) diversity of the taxonomic focus in the scientific literature and EIARs (c). Only one taxonomic group was found to have been studied in our literature survey for the years 2000, 2001, 2002, 2003 and 2010, and thus diversity was zero in those years.

2.5 Discussion

While the global scientific literature and EIARs from Ireland both diversified over time in relation to the taxa and range of impacts included, this diversification took place much earlier and more comprehensively in EIARs. Both datasets were, however, similar in the taxonomic groups that were most or least frequently included; birds were

the most frequently included group in both datasets, while invertebrates and herpetofauna were included least frequently. This adds support to our initial expectation that if a taxonomic group or impact is well-studied in the literature, it will also be well-represented in ecological surveys for EIARs. Conversely, where a group or impact remains understudied in the literature over time, it also tends to receive less attention in ecological surveys.

Our finding that EIARs from the early 2000s had greater taxonomic coverage than the scientific research at that time indicates that there are other drivers of inclusion for ecological surveys in EIARs. As well as scientific research, legislative requirements, ecological knowledge, and established survey guidelines are also key drivers of inclusion of species in EIARs (Briggs & Hudson, 2013). Guidelines developed by the Chartered Institute of Ecology and Environmental Management (CIEEM), which are cited in many Irish windfarm EIARs, have been a driver of change since their first publication in 2006 (Drayson et al., 2015). These guidelines make reference to the importance of published scientific information and promote its inclusion in the EIAR process (CIEEM, 2018). Despite this, EIARs often contain survey methodology that is not up-to-date with the standard in the scientific literature (Singh et al., 2020). The reasoning for this could be a lack of standardised methodologies for biodiversity surveys (Gontier et al., 2006), unclear and broad requirements of the EU Habitats Directive (Council Directive 92/43/EEC), and / or resource constraints within the ecological consultancy setting (Treweek, 1996). While both policy and guidelines are influential drivers of taxonomic inclusion in EIARs (Briggs & Hudson, 2013; Drayson et al., 2015), both will benefit from incorporating up-to-date scientific research.

This study is limited somewhat in spatial context in that the EIARs reviewed were restricted to Ireland, though they are compared against the global literature, and restricting our analyses to European scientific literature does not change our conclusions (Fig. A.2). Language barriers, resource constraints, and variable accessibility of EIARs led to us restricting our EIAR review to Ireland. Our search of the scientific literature returned only three studies from Ireland, all of which studied impacts on birds. Even so, the Irish EIARs were more diverse in the number of taxonomic groups included compared to the global literature, which aligns with the

findings of EIAR surveys from both France (Bigard et al., 2017) and England (Drayson et al., 2017). This indicates that differences in taxonomic coverage and diversity between EIARs and scientific research have been underestimated in our reviews. Regions with higher species richness than Ireland, which is relatively low in biodiversity (Cerqueira et al., 2015), may have higher diversity in their EIARs compared to the Irish reports. The EIARs were also more consistent in what was studied over time compared to the global literature, where attention focused primarily on birds and bats, with occasional inclusions of other taxonomic groups. These findings of our scientific literature review are consistent with the taxonomic bias observed more broadly in conservation (Clark & May, 2002; Tkach & Watson, 2023; Toomey et al., 2017), research funding (Mammola et al., 2020; Adamo et al., 2022) and ecology (Bonnet et al., 2002; Culumber et al., 2019). Birds and, to a lesser extent, mammals tend to be over-represented in the literature relative to their species richness, whereas herpetofauna and invertebrates are typically underrepresented (Bonnet et al., 2002; Mammola et al., 2023). Thus, in addition to collision being an obvious and serious impact for the onshore wind energy sector, explaining much of the focus on volant animals, the dominant focus on these groups also reflect in part more general biases in biodiversity research (Mammola et al., 2020; Troudet et al., 2017; Adamo et al., 2022). Reviews have described a minimal reduction in bias over time (Clark & May, 2002; Fabian et al., 2019). This contrasts somewhat with our finding that the taxonomic diversity is rising relatively rapidly over time, particularly compared to EIARs. Regarding impacts studied, publication bias may contribute to the larger volume of scientific literature on direct, obvious mechanisms of impact (e.g., collision) where a significant effect is more likely (Koricheva, 2003; Rosenthal, 1979), compared to indirect impacts (e.g., habitat loss, behavioural changes), which can be harder to identify and attribute to the windfarm specifically, or due to the difficulty in measuring such impacts.

Ireland has greatly increased its reliance on wind energy since its first windfarm development in 1992 (Wind Energy Ireland, 2023), with capacity increasing steadily over time (SEAI, 2023). Thus, it provides a meaningful case study of rapid increase in demand for EIARs over the past twenty years. Growing demand for wind energy has led to the expansion of windfarms into a range of habitat types across Ireland (Hallan &

González, 2020), and this provides opportunity for new species that may need to be included in EIARs. Variation in taxonomic diversity across years in EIARs could be representative of the environmental context in which approved windfarms were set, and the changes in such contexts over time. Additionally, Ireland is a member state of the EU, and must, therefore, follow legislation and directives regarding planning permission for large developments, which applies to all member states. The EIAR process is also a common development assessment globally (Morgan, 2012). There are, however, some results from the EIARs that reflect the context of the study's setting. For example, vegetation was frequently surveyed in EIARs to provide landscape context for the EIAR, rather than to examine potential impacts to vegetation caused by infrastructure unless it was specified explicitly for a species present within the proposed development listed on the Floral (Protection) Order 2022 (S.I. 235/2022) or was within a habitat listed in Annex I of the Habitats Directive (Directive 92/43/EEC). This further highlights how, if an impact is not obvious, it can be overlooked even if there is scientific evidence for impact (Urziceanu et al., 2021).

Legislation can, however, play an important role in inclusion in impact assessment (Bigard et al., 2017). We could not account for the legal status of species in our data analysis due to the broad taxonomic groupings used, and the difficulties in quantifying the number of species of certain groups in Ireland (e.g., invertebrates). However, we can see that our findings of birds and mammals occurring more frequently in EIARs reflects the proportions of legal status across Ireland's 232 protected species (European Environment Agency, 2020). Birds had the highest proportion of protected species, accounting for 74% of protected species in Ireland, primarily due to the impact of the Birds Directive (Directive 79/409/EEC) (European Environment Agency, 2020). Mammals had the next greatest share of protected species (14%), and all of Ireland's nine bat species are protected under the Habitats Directive (Directive 92/43/EEC) (European Environment Agency, 2020). Invertebrates, specifically arthropods, account for less than 1% of Ireland's protected species (European Environment Agency, 2020). This aligns with our finding that invertebrates are typically not represented in EIARs. Other studies have documented similar gaps between scientific information, policy, and application (Bigard et al., 2017; Knight et

al., 2008). Our findings are not, therefore, unique to Ireland, and similar outcomes would be expected in other countries.

Though we quantified the breadth of taxonomic focus in EIARs, we could not quantify robustly how the survey effort for each taxonomic group specifically changed over time. This was because EIARs varied in consistency of reporting on survey effort and, in many cases, survey effort could not be determined. Though taxonomic coverage of EIARs increased over time, the survey methodologies employed remained largely consistent (Fig. 2.2b). Mammals and invertebrates, for example, remained primarily surveyed using opportunistic methods, even as the number of surveys increased. Bats were the exception, as early reports primarily surveyed through scoping. Legislative protection and increasing knowledge of their ecology in Ireland (Roche et al., 2007) has led to increased employment of field-based assessment methods. Additionally, the use of technology is becoming an increasingly popular method of surveying elusive species such as bats (Gibb et al., 2019; Stephenson, 2020), and continuous improvement of open-source identification software (e.g., BirdNET; Kahl et al., 2021, NEAL; Gibbons et al., 2023) can improve the implementation of surveys for such species in planning assessments.

Scientific understanding has been described as a limiting factor for improving ecological EIARs (Briggs & Hudson, 2013; Drayson et al., 2017). Similar relationships, described as the “science-practice gap”, have been well-documented in other areas, including landscape ecology (Opdam et al., 2001), land management (Burbidge et al., 2011), and nature conservation in general (Knight et al., 2008). Many tools, frameworks, and methods have been suggested to better incorporate science into practice and make it more accessible for practitioners (Bertuol-Garcia et al., 2018; Cooke et al., 2021). Such recommendations are, however, from practitioner’s perspectives, often ineffective or unproductive (Hulme, 2014). In the context of our study, the general perception of the science-practice gap emphasises that the link between scientific research and planning assessments is a one-way relationship, in which scientific findings form the basis for inclusion in ecological planning assessments. In this view, research that is not disseminated to a large audience or that is difficult to access will not be incorporated into EIARs easily. Instead, taking a

different perspective and approaching the knowledge transfer between scientific information and EIARs as a two-way feedback loop can enable the identification of gaps in both knowledge and implementation, ultimately aiding alignment of both. Such a two-way view, in which science informs inclusion in planning assessments and planning surveys reciprocally inform research needs, could contribute to overcoming the “science-practice gap” by encouraging open communication between researchers and practitioners and facilitate knowledge sharing (Gorman et al., 2024). A two-way link is beneficial to acknowledge both general gaps but also specific mechanisms of impact. Often impacts on a taxonomic group tend to be neglected when the mechanism behind a potential impact is cumulative, nuanced, or less discernible (Gontier et al., 2006). For example, collisions tend to be obvious because there is often direct evidence attributable to the presence of wind turbines. However, impacts such as behavioural change or changes to reproductive output are more nuanced and can often have numerous factors contributing to them. Going further, the development of a two-way link between researchers and practitioners can contribute to going beyond understanding impact into developing and co-designing solutions to mitigate and avoid potential impacts to enhance the co-existence of biodiversity and wind energy.

Developing and maintaining a two-way link between scientific research and practitioner action will require transparency, data-sharing, and open communication. Barriers to maintaining such relationships have been identified in similar fields and include the inaccessibility of scientific publishing to practitioners, while other formats of dissemination are often not acknowledged in scientific publishing culture (Fabian et al., 2019). Additionally, there is often little time available for both practitioners and scientists to assign to communicating and sharing information in current formats and, according to practitioners, scientific research often does not align with their actual knowledge requirements (Cvitanovic et al., 2016). More inclusive collaboration between industry, practitioners, and scientists is clearly needed. An open, accessible platform for knowledge sharing where both scientists and practitioners can upload and share knowledge and findings would aid bridging the communication gap. Currently, there are databases for both scientific research on wind energy and wildlife impacts (e.g. American Wind Wildlife Institute; Tethys) and for practitioner knowledge

on the topic (e.g. CANWEA), highlighting that there is a willingness to participate in knowledge sharing. Currently, however, these and similar initiatives are separate (i.e., either for EIAR data only or scientific articles) and not aligned, furthering the one-way perspective on scientific research and practitioners' knowledge. Combining databases and making data-sharing efficient and standardised could provide an essential connection between all parties and ensure knowledge sharing is a two-way link. Such a platform could also support decision-support tools and host communication platforms so that knowledge gaps are identified, and research needs met. Given that legal protection status is a key driver of inclusion in EIARs, policy can play an important role in contributing to solving this science-practice gap.

Ultimately, our study shows that the scientific literature and EIA practice have both diversified in the breadth of their taxonomic coverage and impacts included over time but are not yet fully coherent. Taxa and impacts that were least prominent in both groups are at risk of remaining understudied or undetected over time. Identifying knowledge gaps through enhanced collaboration between scientists, practitioners and stakeholders is essential to ensure correct mitigation is in place to ensure meaningful and impactful solutions that simultaneously address both the biodiversity and climate crises.

Chapter 3 Ecosystem accounting for biodiversity on windfarms

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Author contribution: EK, ID, JCS, and YB conceived the ideas and designed the methodology; EK collected and analysed the data; EK led the writing of the manuscript. All authors contributed critically to the drafts and gave final approval for the publication.

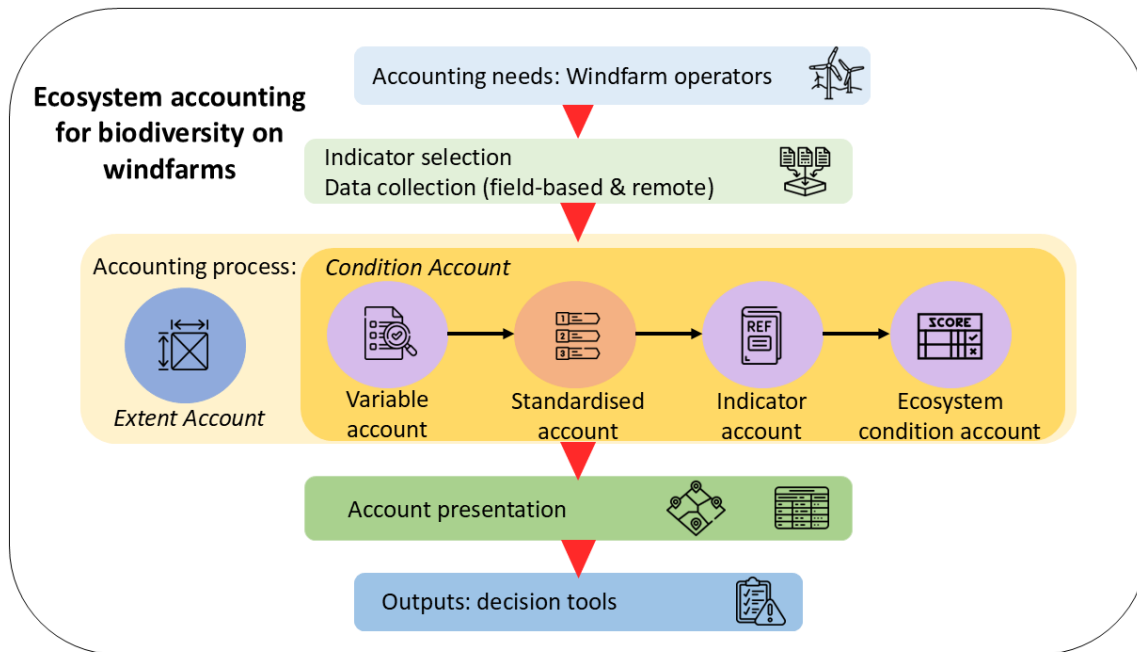
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3.1 Abstract

To ensure that climate action and biodiversity obligations are met, the rapid growth of the renewable energy industry must be accompanied by tools to monitor, protect and enhance biodiversity surrounding renewable energy infrastructures. The United Nations System of Environmental Economic Accounting–Ecosystem Accounting (SEEA–EA) provides a statistical framework to measure and document positive and negative changes in ecosystem extent and quality. Using a spatially explicit approach, this framework can provide industries with a standardised, measurable, verifiable, and reportable approach to ecosystem management. However, accounting for biodiversity, particularly at smaller site-specific scales, is underdeveloped in the SEEA–EA. Here, we use the SEEA–EA to develop biodiversity-themed ecosystem extent and condition accounts for onshore windfarms and provide a management tool for operators, enabling them to understand the extent of the habitats present, and their ability to provide resources for biodiversity. We added a new step into the condition account process to facilitate the incorporation of variables for biodiversity. Application of the methodology highlighted considerable biodiversity potential of some habitats, particularly peatland, marsh, and freshwater systems. The spatially explicit methodology clearly identified locations where improved management could enhance biodiversity. Implementing the SEEA–EA framework is standardised and repeatable, and its flexibility allows for accounts to be built at site-scale for a variety of land-uses and habitats. The development of biodiversity-themed

ecosystem accounts for the onshore wind energy industry demonstrates the utility of the SEEA–EA to inform local management decisions across a sector which is critical for climate change mitigation.

3.1.2 Graphical Abstract



3.2 Introduction

Ecosystem accounting is used to assess ecosystems and their ability to provide services in a format that is similar to financial accounts (Edens et al., 2022). This approach can facilitate the systematic incorporation of natural assets into policy and decision-making, in particular by those who may not be familiar with ecological vocabulary (McGrath & Hynes, 2020). The first framework for the System of Environmental Economic Accounting (SEEA) was published in 1998 following the emergence of environmental accounts over the preceding decade (Edens & Hein, 2013). The United Nations System of Environmental Economic Accounting (UN SEEA) has, since 2012, become the international standard framework for ecosystem accounting. The SEEA–Ecosystem Accounts (EA) has been ratified as a statistical standard since 2021. The SEEA–EA differs from other accounts within the SEEA framework in that it uses a spatial approach to compile accounts, enabling combination of multiple spatially-explicit layers from geographical, environmental, and economic datasets into a single account. Biodiversity, defined as “the variability

among organisms from all sources, encompassing all ecosystems and ecological systems to which they are part, including diversity within species, between species, and of ecosystems” (United Nations, 2011), presents a particular challenge for ecosystem accounting (Grover et al., 2023).

Ecosystem accounts can be compiled at a range of scales and are often used as a tool to measure changes in natural capital. Natural capital is the stock of natural resources, from which flow goods and services that have benefit and value to humanity (Costanza & Daly, 1992). Ecosystem accounting has primarily been conducted at national and regional scales to measure the stocks and flows of natural capital through various systems and sectors (CSO, 2018; Farrell et al., 2021). The focus on national scale accounting can be attributed to the explicit, and intentional, link between the SEEA and the System of National Accounts, which allows the extension of national economic accounts to incorporate natural capital (Bagstad et al., 2021). Existing ecosystem accounts created using the SEEA–EA methodology have been developed for several countries (Bond et al., 2013; Hein, et al., 2020b) and even at the EU-scale through the Integrated system for Natural Capital and ecosystem services Accounting (INCA; La Notte et al., 2022). These accounts have focused primarily on provisioning and regulating services derived from natural capital stocks such as carbon, water, and timber (European Commission et al., 2019, 2021). Given the European Union’s new Corporate Sustainability Reporting Directive (CSRD; Directive (EU) 2022/2464 of the European Parliament), many companies will now be required to incorporate sustainability and environmental matters into their annual reports, including measurements of the impact of their operations on natural capital. Ecosystem accounting approaches, such as the SEEA–EA, are, therefore, increasingly applied at local-, site-, and corporate-level scales. There are benefits to businesses of utilising ecosystem accounts, as they can be applied in decision-making, reporting, and planning (Ingram et al., 2022) and are repeatable to document change through time. However, the application of ecosystem accounts at site-level scales and the complexities of biodiversity accounts using the SEEA–EA are not yet well developed.

Debate has addressed whether biodiversity should be incorporated as an ecosystem service, an output, or as a theme for accounts (Edens & Hein, 2013).

Elements of biodiversity are found throughout the ecosystem services concept, and how it has been included in previous work varies greatly (Mace et al., 2011). The complexity in defining biodiversity within the ecosystem services concept has compounded confusion as to biodiversity's place within ecosystem accounting (Reyers et al., 2012). This has, in turn, slowed development of biodiversity accounts compared to other ecosystem accounts (Reyers et al., 2012), and there remains considerable uncertainty in structuring biodiversity within ecosystem accounts (Remme et al., 2016). Mace et al. (2011) recommended using the Convention on Biological Diversity's definition of biodiversity (United Nations, 2011), placing biodiversity primarily as an ecosystem service.

The SEEA-EA framework comprises five core accounts – extent, condition, physical ecosystem service, monetary ecosystem service, and ecosystem asset accounts (United Nations, 2021), which can be further divided into two types – stock and flow accounts (Edens et al., 2022). Stock accounts, which consist of ecosystem extent and condition, are the most developed for biodiversity-themed accounting (Edens et al., 2022). While guidance is available on monetary (flow) accounts, the difficulty in quantifying the value of nature has limited progress for the creation of biodiversity-themed monetary accounts (Edens et al., 2022; Hein et al., 2015). Moreover, for biodiversity, there is debate on whether condition accounts should be used to infer the ecosystem services accounted for, or whether the services should be selected first and the condition or capacity accounts built for that service (Faccioli et al., 2023; Martini et al., 2024).

Often, the first ecosystem account to be built is the extent account. This is defined as the spatial measurement of an ecosystem asset, which is the primary unit that comprises the accounts (United Nations, 2021). In terms of biodiversity, a habitat classification is typically used. Ecosystem condition then measures the quality of an ecosystem asset in terms of its abiotic and biotic characteristics relative to a reference condition (United Nations, 2021). Ecosystem condition is, in general, the second account developed, building on the extent account. Extent and condition can be used to determine targets for management of habitats that subsequently affect ecosystem services (Maseyk et al., 2016). Characteristics of the habitat are systematically

organised and transformed through a series of stages to result in an index value that acts as a proxy for the quality of the asset. A range of criteria need to be met to select indicators that act as robust proxies of the condition of the focal ecosystem (United Nations, 2021), indicators need to be spatially explicit at the selected scale (Rendon et al., 2020), sensitive to environmental change (Czúcz, et al., 2021b), and relatively easy to measure (Czúcz, et al., 2021b). It is also important that there is a clear evidence-based link between indicator and the service the account is compiled for (Czúcz, et al., 2021b; Rendon et al., 2020). Defining this link is, however, often difficult (Remme et al., 2016). As biodiversity has been poorly aligned within ecosystem accounting (Mace et al., 2011), difficulties in selecting suitable indicators can be compounded. Once potential characteristics have been identified, an additional challenge is sourcing a dataset or finding a way to measure those characteristics in the field (Faccioli et al., 2023). Accessibility of data is a major limitation in many use-cases of ecosystem accounting (Czúcz, et al., 2021b; Feld, et al., 2009; Remme et al., 2016; Rendon et al., 2020). In addition, indicators need to have appropriate reference conditions relative to the ecosystem and service for which the accounts are being developed (Martini et al., 2024). These can be difficult to source, particularly for heavily modified ecosystems (Rendon et al., 2020).

Accounting for biodiversity at the site-scale has had limited implementation and uptake due to perceived barriers (van Putten et al., 2021). Ecosystem accounting has undoubted potential to become a useful management tool for industries or corporate businesses to make informed decisions or predict change to areas under their management. Developing accounts at the site scale can structure the collation of detailed ecosystem accounts in a standardised format that aligns with reporting requirements. The flexibility of the SEEA-EA approach, while maintaining its standardised format, is a key benefit of the framework, especially in how it integrates across multiple land-uses and industries.

Use of standard National Classification system(s), such as the Irish National Land Cover Map (Lydon et al., 2023), is an important component of building accounts, as they are typically used in the design and planning of surveys and align with national datasets. The SEEA-EA recommends the use of the International Union for

Conservation of Nature (IUCN) Global Ecosystem Typology (Keith et al., 2022), which could also be useful in site-level accounting, particularly for multinational companies that may need to account for multiple sites across many countries. Therefore, the ability to switch between different classification systems would be beneficial when conducting site-level accounting. However, we need to understand whether, and how, using different classification systems influences the accounting output. Additionally, an important use of ecosystem accounting for industry is as a management tool or as a basis to develop further management tools. It is important that the accounts developed can identify differences in condition indices among different habitat types, which would help prioritise management actions and facilitate informed decision-making.

The onshore wind energy industry in Ireland provides an excellent case study to test the flexibility of the SEEA–EA, as wind turbines are co-located across many land-use types and habitats on the island (Hallan & González, 2020), and operators require tools to facilitate management decisions incorporating the biodiversity that surrounds their wind turbines (Gorman et al., 2023). Further, previous case studies have demonstrated the ability to develop ecosystem accounts at larger scales in the Irish landscape (Farrell et al., 2021) and planning for onshore windfarms in Ireland typically utilises a specific national habitat classification system (Fossitt, 2000), which provides the opportunity for comparison against the SEEA–EA recommended typology (Keith et al., 2022).

Here, we develop a workflow to specifically address the challenges of scale, incorporation of biodiversity as an ecosystem service, and data availability for thematic natural capital stock accounts using the SEEA–EA framework. We apply this framework to build site-scale ecosystem accounts for seven exemplar windfarms across Ireland. The accounts were built primarily to identify areas of natural capital that enable biodiversity enhancement or protection across multiple habitats and land-uses. We use our results to compare ecosystem accounts based on two different standardised habitat classification systems (that is, the Fossitt classification system and the SEEA–EA-recommended IUCN global ecosystem typology) to understand the extent to which these different systems affect condition scores. We compare

condition scores among habitats to assess the ability of the SEEA-EA to measure biodiversity as an ecosystem service. Finally, we discuss the limitations of our approach, and of ecosystem accounting in general, and make recommendations on the application of ecosystem accounting at local-scales.

3.3 Materials and Methods

3.3.1 Indicator Development

In the SEEA-EA, the Ecosystem Condition Typology (ECT) is “*a hierarchical typology for organising data on ecosystem condition and characteristics*” (Czúcz, et al., 2021a). This typology was developed to standardise and structure the selection of variables included in the ecosystem accounts. To select variables, relevant literature on biodiversity indicators and ‘scorecards’ for similar habitats were reviewed. The variables selected should have a clear biophysical link to the goal of the accounts being developed (Czúcz et al., 2021a). Here, our goal was to develop biodiversity-themed accounts that can be used to inform management decisions for the Irish wind energy industry. Thus, the variables need to have a biophysical link with the potential for the ecosystem to provide habitat for biodiversity.

A literature review of biodiversity indicators and a review of available datasets was conducted to provide a list of 34 potential variables that could be included in the condition account. A workshop was held with five independent expert ecologists to discuss the suitability of potential indicators prior to inclusion. In the ECT, there are typically six classes (ranging from physical state to landscape/seascape state) across three groups (that is, Abiotic, Biotic and Landscape). All potential indicators for the Biotic grouping, ‘Functional state characteristics’, could also be seen as other characteristics (structural or compositional state). Thus, no functional indicators were selected for the biodiversity accounts.

For variables that require data to be collected in the field, index scorecards were developed (Appendix B1). The scorecards follow a similar format to others developed in Ireland [e.g., Wild Atlantic Nature Project (2024), Pearl Mussel Project (2018), FarmPEAT (2024); McLoughlin et al., 2020]. The scorecards needed to be practical for

use in the field and also for stakeholders to be able to easily use and input the data into the accounting framework. Therefore, the condition variables in the score cards needed to be easy to record, easy to interpret, standardised across habitats of the same type, and in units that are easy to follow.

3.3.2 Implementing the SEEA-EA

The first accounts built were the extent accounts using area of each habitat. The accounts were grouped based on the habitat type. As we only gathered data from one set of surveys, the standard SEEA-EA extent account format was modified to account for one observation period rather than changes over time. That is, accounts were created using only ‘opening values’ as no ‘closing’ data were collected.

In the SEEA-EA framework, developing ecosystem condition accounts is a three-step process producing three accounts: the variable account, indicator account, and ecosystem condition account. The variable account is produced from inputting biodiversity characteristics. An additional step that is not in the SEEA-EA framework was added for the biodiversity accounts produced here, in which the variable was rescaled to an interval index between 0-3, where 0 indicates poor condition and 3 indicates best-attainable condition (Fig. 3.1). The addition of this step was to address directionality, non-linearity of the relationship of the variable to condition and, specifically for biodiversity accounts, issues surrounding specificity required of reference conditions (Rowland et al., 2021). These variables were then scaled using reference conditions to produce the indicator value I , as:

$$I = \frac{(V - V_L)}{(V_H - V_L)} \quad \text{eqn. 1}$$

where V is the standardised variable value, V_L is the lower reference value, and V_H is the upper reference value. The indicator value I is therefore a continuous measure between 0-1, with zero indicating poor condition and one indicating best attainable condition.

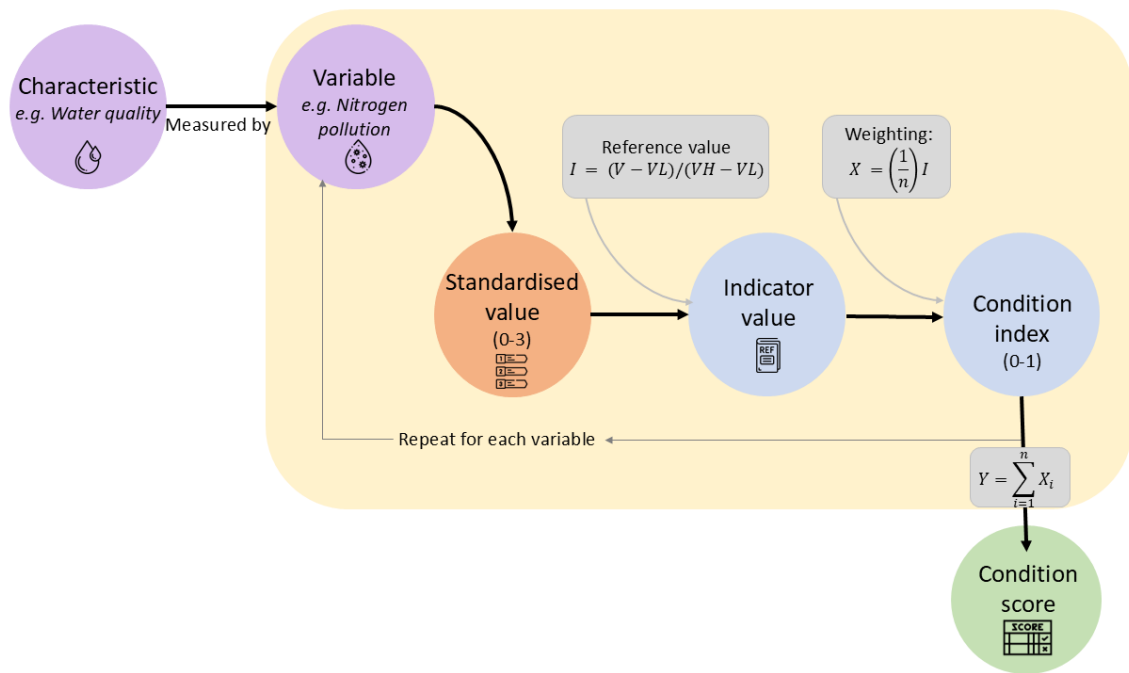


Figure 3.1. Outline of steps involved in developing a condition score for condition accounts. Characteristics, such as water quality, are measured by variables, which are quantitative metrics to describe characteristics. They must be unambiguous and have well-defined measurement units. Variables are then rescaled to a standardised value on a 0-3 scale, allowing for non-linear relationships between variables and condition. Following this, they become indicators through rescaling against a reference value (V_L : lower reference value, V_H : upper reference value). They are then weighted depending on the relative importance of each indicator to the overall condition score (N : total number of indicators). This process is repeated and the condition indices for each variable are summed to produce the condition score for the ecosystem asset.

The SEEA-EA proposes several possible reference conditions that can be used for anthropogenic or highly-modified ecosystems. Best-attainable condition was selected as it aligned most closely with the goal of the ecosystem accounts for our purposes. That is, to provide good management tools in a highly modified environment. Once variables were scaled against the reference condition, they become indicators in an indicator account. The final condition account was created from aggregated condition indices, with an equal weighting (together equating to one) across the Ecosystem Condition Typology classes. Following the SEEA-EA recommendations, equal weighting was used across all ecosystem classes. Condition accounts were not created for buildings and artificial surfaces (BL3), spoil and bare ground (ED2), or urban and industrial ecosystems (T7.4) of the Global Ecosystem Typology (Keith et al., 2022) owing to the lack of biodiversity indicators for these anthropogenically-created habitats. Interactive maps displaying condition indicator

scores were created using the ‘mapview’ (Appelhans et al., 2023) and ‘leaflet’ (Cheng et al., 2024) packages in R version 4.2.2 (R Core Team, 2022), and made available via the ‘shiny’ (Chang et al., 2024) package (See supplementary material for an example interactive map).

3.3.3 Site Selection

Our case study windfarm sites were selected as representative of onshore windfarms across Ireland and the habitats and land-uses in which they are typically co-located. Seven sites in the Republic of Ireland were used as exemplars for which stock accounts were developed. Land-use in Ireland is dominated by grasslands, followed by peatlands, croplands, and forest (CSO, 2023), and this was replicated in site selection. Windfarm sites were also selected to represent a mixture of sizes (5 – 36 turbines; 65 – 2187 ha) and ages of operation (established 2002 – 2018). There are two common land management types for windfarms in Ireland – ‘wholly-owned’ and ‘herringbone lease’. In the case of wholly-owned windfarms, the entire site, incorporating the windfarm infrastructures and surrounding land, are owned and managed by the windfarm operator. For sites under herringbone lease, the area under management by the operating company is the footprint of the windfarm infrastructure, whereas the surrounding land is typically owned by separate landowner(s).

Environmental Impact Assessment Reports (EIARs) and other relevant ecological information were provided by windfarm operators. Sites were then chosen to maximise representation of the range of windfarms in operation in Ireland (that is, in terms of habitat type and land use, lease type, age, number of turbines, and geographic location).

3.3.4 Data collection (*Extent & Condition*)

Initial site visits were conducted between June and September 2022. Prior to this, habitat maps were drafted in ArcGIS Pro (ESRI, 2023) using boundary and turbine location data provided by the site operators and aerial imagery to assign broad habitat types. Onsite, a walkover survey was conducted, in which each habitat polygon was mapped if greater than the minimum mappable area of 400 m² (following Smith et al., 2011). The polygon was assigned to an Ecosystem Functional Group (EFG) from the

IUCN's Global Ecosystem Typology (Keith et al., 2022) (Table B.1) and a Fossitt Code (Level 3; Fossitt, 2000), a standard scheme for describing habitats in Ireland (Table B.2). Some habitats did not fit into one Fossitt Code so were placed into a combination of two Fossitt habitat types. Alignment between the habitat classifications is outlined in the Appendix (Table B.3). Land use was determined based on the dominant habitat type identified on the site, e.g., agriculture, commercial forestry, and peatland. Habitats were mapped in the field directly onto the preliminary maps using ERSI ArcGIS Field Maps software (ESRI, 2023). Data were input into ArcGIS Pro and the habitat polygons were digitised and the area of each was extracted, as was the total area of each habitat type (Fig. 3.2).

Both remote and field-based data collection were used to build condition accounts. The condition scorecards were used in the field to collect data on habitat condition where applicable. Remote data sources were manipulated and analysed in ArcGIS Pro to extract data relevant to the specific habitat area.

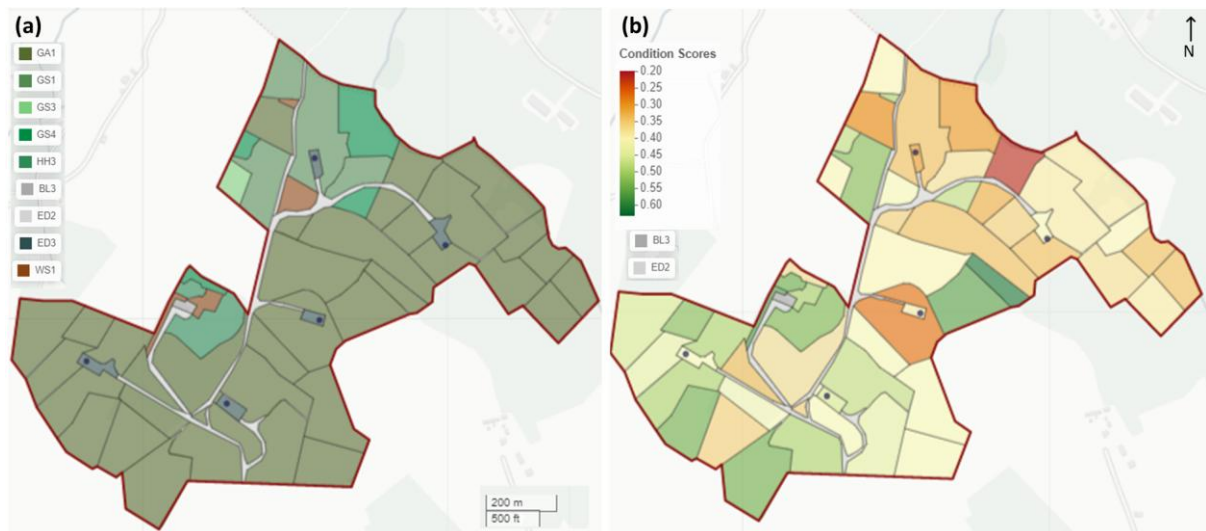


Figure 3.2. Extent map of habitats at an onshore windfarm, mapped using the 'Guide to Habitats in Ireland' (Fossitt, 2000) (a). The land-use for this site was agriculture, as the site is dominated by improved agricultural grassland (GA1). Condition scores mapped across individual polygons for the same onshore windfarm (b). Areas to the south-west of the site tended to have greater condition scores than those in the north-east. Blue points represent the turbine locations on both maps.

3.3.5 Statistical Analyses

Generalised linear mixed models (GLMMs) were used to assess how condition scores varied among habitat types to understand the ability of the SEEA-EA to measure

biodiversity as an ecosystem service. The categorical main fixed effects comprised land use (defined as agriculture, commercial forestry, or peatland) and habitat type. Condition score was the response variable, and windfarm site was included as a random effect. A null model, which contained no fixed effects, was used to determine the random effect structure. Separate models were built using the IUCN habitat type and Fossitt Level 3 habitat type.

Weighted Effect Coding (te Grotenhuis et al., 2017) was used to assess whether using alternate classification systems affects condition scores. Weighted Effect Coding was used to test the difference among factor level means, weighted to account for differences in sample sizes among factor levels and to set a specific reference value for the model. Model diagnostics were run using the DHARMA package (Hartig, 2022), and data visualisation used ggplot2 (Wickham, 2016). Akaike Information Criterion (AIC) values were used to compare between GLMMs for each habitat type (Bolker, 2008) to understand whether the model outputs varied between the two different habitat classification systems. Given that the models differ in fixed effects while using the same response variable, the difference in AIC is due to the different explanatory power of the fixed effects in our models.

3.4 Results

3.4.1 Indicator Selection

Thirty-two indicators were selected for inclusion in the ecosystem accounts (Table B.4). Of these, seven were field-based, 13 were remote, and the remainder (12) combined field-based and remote elements. The indicators selected were not evenly distributed across either the SEEA–EA Ecosystem Condition Typology (Fig. 3.3a) or habitat type (Fig. 3.3b).

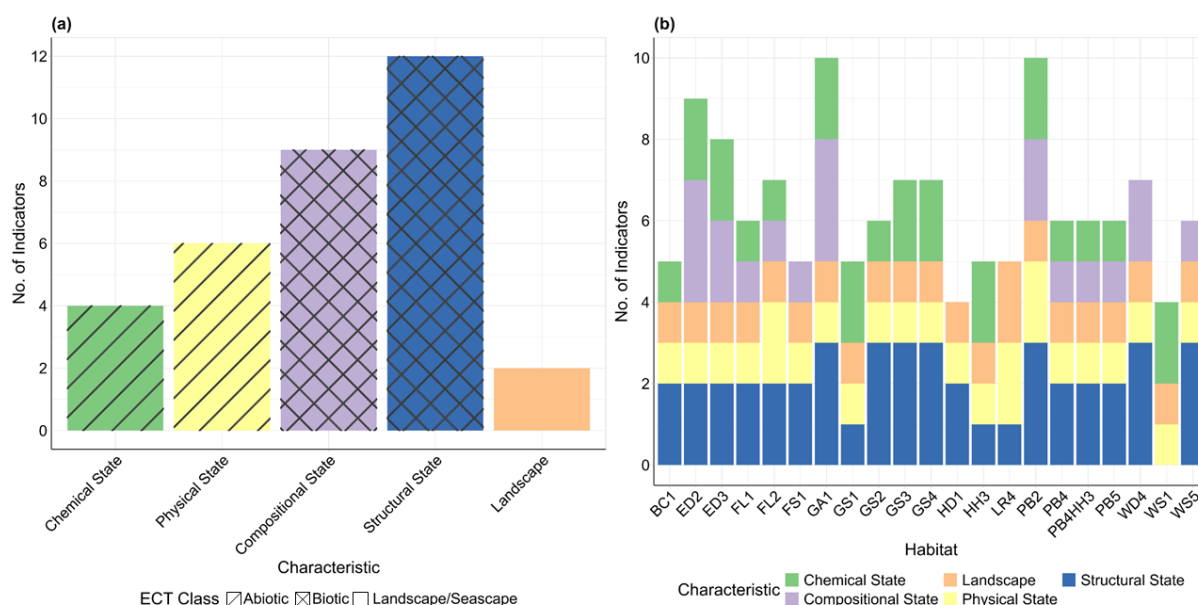


Figure 3.3. Distribution of indicators across the SEEA-EA's Ecosystem Condition Typology characteristics (a) and across the Fossitt (2000) habitat types (where BC1 – Arable crops; ED2 – Spoil and bare ground; ED3 – Recolonising bare ground; FL1 – Dystrophic lakes; FL2 – Acid oligotrophic lakes; FS1 – Reed and large sedge swamps; GA1 – Improved agricultural grassland; GS1 – Dry calcareous and neutral grassland; GS3 – Dry-humid acid grassland; GS4 – Wet grassland; HD1 – Dense bracken; HH3 – Wet heath; LR4 – Mixed substrata shores; PB2 – Upland blanket bog; PB4 – Cutover bog; PB4HH3 – Cutover bog & wet heath mosaic; PB5 – Eroding blanket bog; WD4 – Conifer plantation; WS1 – Scrub, WS5 – Recently-felled woodland) (b) identified on Irish windfarm sites (codes on x-axis), and their ECT characteristics (colours as per (a)). Structural indicators (dark blue) occurred most frequently across habitats, excluding WS1 (Scrub). GA1 (Improved agricultural grassland) and PB2 (Cutover bog) had the highest number of indicators used to assess condition. Full definitions of each Fossitt (2000) habitat code can be found in Table B.2.

3.4.2 Extent Accounts

The seven exemplar windfarm sites were categorised into broad land use types that included commercial forestry, upland blanket bog and heath, and agriculture (predominantly pasture with some tillage). Across the seven sites there were 27 habitats identified using the Fossitt level 3 classification (Fossitt, 2000). Improved agricultural grassland (GA1), and conifer plantation (WD4) were the two most frequently occurring habitats. Using the IUCN Global Ecosystem Typology 2.0 (Keith et al., 2022), 12 habitat types were identified at the Ecosystem Functional Group level. Sown pastures and fields (T7.3), urban and industrial ecosystems (T7.4), and boreal, temperate, and montane peat bogs (TF1.6) were the most frequently occurring habitats identified using the IUCN typology.

3.4.3 Condition Accounts

The final dataset included 592 condition scores for individual habitat polygons across the seven sites. There were 24 Fossitt Level 3 habitat types that condition was accounted for, and 11 IUCN Ecosystem Functional Group habitat types. Condition scores ranged from a minimum 0.17 to a maximum score of 0.89, with a mean score of 0.47 (from a possible range of 0 to 1, where zero represents poor habitat condition for biodiversity and 1 represents good habitat condition for biodiversity). Across the land use types, sites with commercial forestry as their land use had the highest average condition score (Fig. 3.4a, Table B.5).

3.4.4 Differences Among Habitats

Using the IUCN Global Ecosystem Typology 2.0 (Keith et al., 2022) classification, condition scores varied at the Ecosystem Functioning Group level (Table B.6). In general, small permanent freshwater lakes (F2.2), annual croplands (T7.1), permanent marshes (TF1.3), and boreal, temperate, and montane peat bogs (TF1.6) had higher than average condition scores (Fig. 3.4b). In contrast, derived semi-natural pastures and oldfields (T7.5), cool temperate heathlands (T3.3), urban and industrial ecosystems (T7.4) and sown pastures and fields (T7.2) had lower than average condition scores.

Condition scores also varied by habitat type across Fossitt habitat classifications (Table B.7). In general, arable crops (BC1), dystrophic lakes (FL1), reed and large sedge swamps (FS1), upland blanket bog (PB2), cutover bog (PB4), cutover bog/wet heath (PB4HH3), and recently-felled woodland (WS5) had higher than average condition scores. In contrast, recolonising bare ground (ED3), acid oligotrophic lakes (FL2), depositing lowland rivers (FW2), improved agricultural grassland (GA1), dry calcareous and neutral grassland (GS1), dry meadows and grassy verges (GS2), dense bracken (HD1), and scrub (WS1) had lower than average condition scores (Fig. 3.4c).

Condition scores did not significantly vary with land use type (Fig. 3.4a). Moreover, the explanatory power of both the IUCN Ecosystem Typology (Fig. 3.4b, Table B.8) and the national Fossitt system (Fig. 3.4c, Table B.9) in describing condition scores was similar (AIC for models based on the IUCN Ecosystem Typology: df 15, AIC

-871.4; AIC for Fossitt Habitat Classification: df 27, AIC -873.4). The difference in AIC values of the model (ΔAIC) was <2 , indicating that the two models have similar support from the data, and were similar in their output (Bolker, 2008).

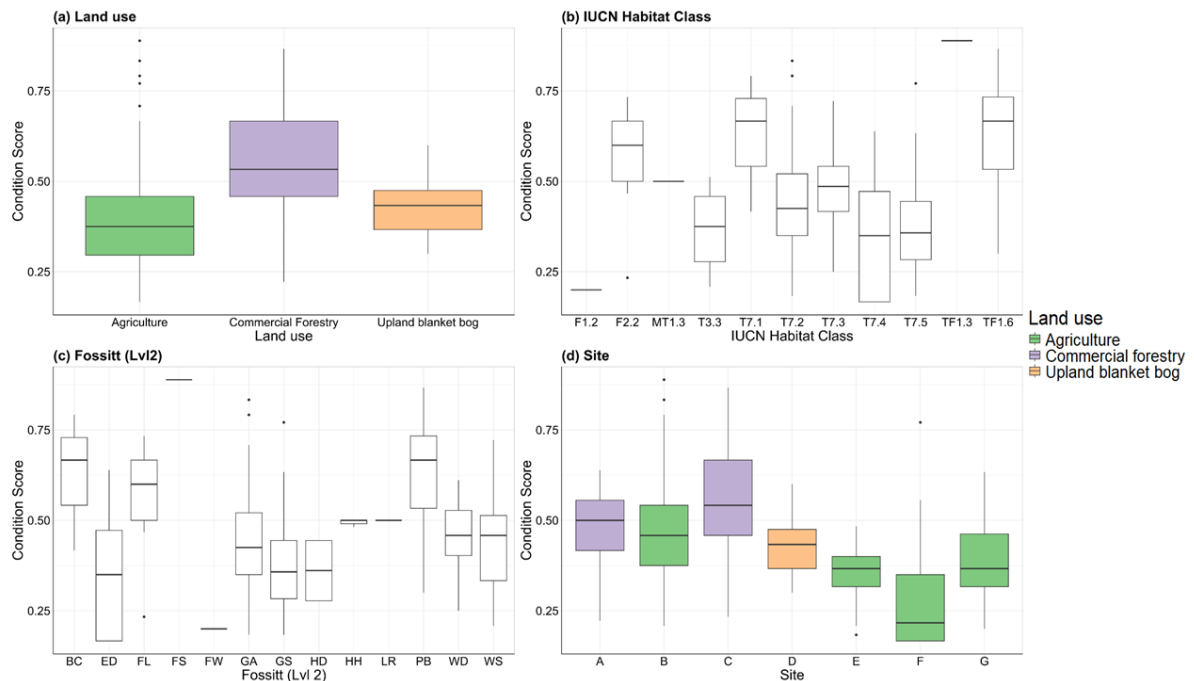


Figure 3.4. Condition scores for wind farm land use type (a), IUCN habitat type (where F1.2: Permanent lowland rivers; F2.2: Small permanent freshwater lakes; MT1.3: Sandy shorelines; T3.3: Cool temperate heathlands; T7.1: Annual croplands; T7.2 Sown pastures and fields; T7.3: Plantations; T7.4: Urban and industrial ecosystems; T7.5: Derived semi-natural pastures and oldfields; TF1.3: Permanent marshes; TF1.6: Boreal, temperate and montane peat bogs) (b), Fossitt Level 2 (BC: Cultivated and build land; ED: Disturbed ground; FL: Lakes and ponds; FS: Swamps; FW: Watercourses; GA: Improved grassland; GS: Semi-natural grassland; HD: Dense bracken; HH: Heath; LR: Littoral rock; PB: Bogs; WD: Highly modified/non-native woodland; WS: Scrub/transitional woodland) (c), and windfarm sites (anonymised; AG: Agriculture; CF: Commercial forestry; BB: Upland blanket bog) (d).

3.5 Discussion

We developed thematic ecosystem accounts for seven windfarm sites across Ireland, introducing an additional step into the SEEA-EA framework to incorporate information on data uncertainty, non-linearity, and directionality. Our accounts provide an example application of the SEEA-EA framework as a tool for local-scale biodiversity monitoring and highlight the adaptability of the method. We also demonstrate the flexibility of the SEEA-EA, as our results show the interchangeability of using national classification systems [i.e., Fossitt Guide to Habitats in Ireland

(Fossitt, 2000)], with the recommended IUCN Global Typology (Keith et al., 2022), with both systems having similar explanatory power for condition scores. Previous work in ecosystem accounting focused primarily on the regional (e.g. Farrell et al., 2021) and national (e.g. Hein, et al., 2020b) scales. Here we provide a case-study for accounting at the site scale for the purpose of management decision-making. The biodiversity-themed accounts identify the capacity of the ecosystem type to provide habitat or resources for biodiversity. Outputs of the accounts were used to identify preferable habitat for biodiversity, or areas requiring greater management input.

Under the IUCN Global Ecosystem Typology (Keith et al., 2022), small permanent freshwater lakes (F2.2), permanent marshes (TF.3), annual croplands (T7.1), and peat bogs (TF.1) had higher than average condition scores. Similar habitat types under Fossitt (2000), a common national-scale typology used in biodiversity surveying in Ireland, also had higher than average condition scores, such as upland blanket bog (PB2), arable crops (BC1), reed and large sedge swamps (FS1). Many of these align with research on biodiversity assemblages as, for example, in Ireland peatlands such as upland blanket bog can support unique assemblages of flora (Sottocornola et al., 2009), birds (Bracken et al., 2008) and invertebrates (Hannigan & Kelly-Quinn, 2014). The further division of peatlands in the Fossitt (2000) classification system demonstrated the ability of cutover peatlands (PB4) to provide habitat for biodiversity (Crowley et al., 2021; D'Astous et al., 2013), albeit slightly lower on average than upland blanket bog (PB2). This highlights an advantage of the national classification system, which includes breakdown of habitats that are more specific, with different reference levels and localised to Ireland, compared to the generalised peat bogs (TF1.6) category in the IUCN typology.

Similar habitat types in the two different classification schemes show a significantly lower than average condition score. Under the IUCN typology these were derived semi-natural pastures and oldfields (T7.5), cool temperate heathlands (T3.3), urban and industrial ecosystems (T7.4) and sown pastures and fields (T7.2) and, under Fossitt (2000), dry meadows and grassy verges (GS2), recolonising bare ground (ED3), improved agricultural grassland, scrub (WS1) and dense bracken (HD1). For pastures and agricultural grasslands (e.g., T7.5, GA1, GS2) and highly modified habitats (T7.4,

ED3), this would be expected, as these areas are often highly altered and can be difficult for species to utilise (Plantureux et al., 2005; Urziceanu et al., 2021, 2024). Comparing the model fit for condition scores showed interoperability between the two classification systems, with similar explanatory power on condition scores. This allows for flexibility in terms of use as a management tool, as the Irish standard classification system is more practical for practitioners, whereas the IUCN classification system would allow greater capacity for international windfarm operators to compare sites across regions.

At the broadest level of classification (land-use), commercial forestry had the highest average condition score, compared to agriculture or commercial bog. The biodiversity potential of commercial forestry plantations has long been debated (Brockhoff et al., 2008) and, despite the overall land-use resulting in the highest average condition score, with the finer grained habitat classifications (Fossitt (2000) Level 3 and IUCN Global Ecosystem Typology EFG) land use did not have a statistically significant effect on the condition scores. Each land-use, that is, commercial forestry, agriculture, and peatlands, can be considered an economically productive land-use in Ireland (Bullock et al., 2012; DAFM, 2021; Ní Dhubháin et al., 2009), and previous research has highlighted that plantation forests can have a higher biodiversity benefit than other productive land-uses, in particular agriculture (Brockhoff et al., 2008). In addition, commercial forestry in Ireland is often located on peat soils due to historic afforestation grants (Bullock et al., 2012), and therefore the commercial forestry sites included here also had peatland and lake habitats within them, which were some of the habitats that had higher than average condition scores (peat bogs TF1.1 in IUCN, and PB2 in Fossitt (2000), and freshwater lakes F2.2 in IUCN). Assessment of the biodiversity value of commercial forestry plantations themselves rather than sites classed as commercial forestry due to the dominance of this land use therefore require a more fine-grained approach.

The SEEA-EA was developed to fit alongside the System of National Accounts (SNA), and facilitate countries accounting for non-economic capital in their national reports (United Nations, 2021). The SEEA-EA framework can also be used for smaller-scale accounting but, due to their design for national level purposes, some

approaches are not feasible at smaller site scales (Faccioli et al., 2023). Non-financial reporting is becoming an increasingly popular exercise at corporate level and, as a UN Statistical Standard, the SEEA-EA is well-placed to facilitate such accounting. However, adaptation is required to tailor accounts to smaller scales. Our results highlight that corporate ecosystem accounts can be compiled at the site-level and can identify of within-site variation that could be used to inform management decisions.

Ecosystem accounting is a data-driven process, and data availability is a limitation of the SEEA-EA at all scales (Hein, et al., 2020a; Vári et al., 2024). This issue is, however, particularly acute at local scales (Faccioli et al., 2023). In developing ecosystem accounts for onshore windfarm sites, many available spatial datasets could not be included, as they were at a resolution too large to cover the range in the site's boundary areas and provide sufficient detail. Thus, 59% of indicators selected required an element of field-based data collection. This was possible in that the sites were accessible, and the indicators used could be gathered by observing the focal ecosystem asset. Scorecards, used to reduce field-based subjectivity, were particularly useful for biodiversity-related indicators, as they allowed for a range of values to be represented by a single value in a 0-3 index. This meant that indicators of biodiversity, such as vegetation structure, were quantifiable and could be included in the accounts. The introduction of a standardisation step was an important addition that increased the feasibility of creating biodiversity-themed accounts. Using a standard 0-3 index also allowed for clarity surrounding directionality of variables, especially more complex variables, where both a high value and low value could be negative for biodiversity.

In the SEEA-EA framework, directionality in the condition accounts is poorly addressed, as there cannot be more than one value for each of the high and low reference values. For example, both low and high values of tree cover density can be negative for biodiversity (Smith et al., 2009). Following the traditional SEEA-EA framework, it would have been difficult to include two values as a lower reference value. By placing both values lower in the standardised index, however, this issue can be overcome, and the variable can still contribute towards the accounts. Ryan et al. (2023) also transformed their data prior to the reference stage of the condition

accounts, using a dimensionless scale. The addition of a 'standardisation step' into the condition accounts allows more flexibility in cases where data may be limited or more complex, which is common for biodiversity indicators.

Biodiversity-themed accounts are the least developed accounts of the SEEA–EA (Remme et al., 2016), as the pace slowed due to debate surrounding biodiversity's place within the ecosystem services concept (Grover et al., 2023; Mace et al., 2011; Reyers et al., 2012). The accounts developed here use habitat-based metrics to infer change in biodiversity (Larsen et al., 2021), through identifying the habitat's capability to support and provide resources. Spatial scale has been described as one of the main barriers to accounting for biodiversity, which led to the development of a guidance paper on the topic (Larsen et al., 2021). Here, the scale of accounting was determined by the windfarm operator's site boundaries. This limited the availability of data, as much remote data are found at a scale of 10 km² [e.g., Biodiversity Maps from the National Biodiversity Data Centre (NBDC, 2024)]. Acknowledging that there is ongoing work to continue to progress accounting for biodiversity (United Nations, 2021), it is evident there is more research needed to allow biodiversity-themed accounts to progress to the extent of other themes, such as carbon and water. A lack of suitable ecological data or indicators of the status or trends in biodiversity is one of the major limitations encountered when developing accounts for onshore windfarms, and in other accounting exercises (Faccioli et al., 2023; Feld, al., 2009). Further, in instances where biodiversity data are available, time-series data to document change may not align with the accounting period. Selecting biodiversity indicators to inform on the status of biodiversity was a challenging process (Hein, et al., 2020a), requiring many iterations and expert consultation.

The ecosystem accounts built here provide a case study of accounts developed for a management purpose. The aim was to develop accounts that could inform on the windfarm's biodiversity capital or potential capital, which could be integrated into management tools such as a risk register (Mace et al., 2015) and biodiversity action plans. There are several decisions that are required when tailoring accounts for industry or specific use cases (Gorman et al., 2024). One over-arching concern in developing accounts for industry is ensuring the process, from development through

to delivery of the accounts, was accessible to non-experts in the onshore wind energy industry. This included selecting indicators that were open-access, or relatively straightforward to collect in the field. Striking a balance between this accessibility and the need for meaningful scientific biodiversity indicators was challenging.

Developing ecosystem accounts is a time-consuming process and accounts built for industry will also need to include time in their framework to facilitate educating those in industry about the process, through workshops or training. Companies' capabilities vary, and some may not have the resources required to continue ecosystem accounting themselves (van Putten et al., 2021). Consultants may, therefore, be required to build accounts on behalf of the industry. Development of accounts for industry can be collaborative, incorporating GIS specialists, environmental experts, ecologists, and asset managers (Bagstad et al., 2021). Multiple people may, therefore, need to be trained on the topic. When developing accounts for industry, the purpose of the accounts must be explicit from the beginning of the process, whether it be for biodiversity capacity, to measure service-flows of natural capital, or to monitor change in natural capital over time (e.g. Keith et al., 2020). Selection of indicators can be the most time-consuming and challenging part of the SEEA-EA process (Czúcz, et al., 2021b). In this case study, development of a scorecard of indicator variables hastened the process of collecting data in the field, especially data that was already rescaled in the additional standardisation step to incorporate biodiversity data. Martini et al. (2024) provide an example of the additional rescaling standardisation step using just remote data for the forestry industry. While the SEEA-EA framework provides comprehensive instructions and amendable spreadsheets that can be used to develop extent and condition accounts, further standardised guidelines, specifically focused on accounting for biodiversity, must be developed to facilitate industry's ecosystem accounting needs (Czúcz, et al., 2021b; Edens et al., 2022). Additionally, another option to improve the feasibility of industries developing ecosystem accounts is to build tools that can automate the accounting process [e.g., ExactR (Gibbons et al., 2024) ARIES (Villa et al., 2009)].

The format of the accounting outputs is another important consideration when developing ecosystem accounts targeted for industry (van Putten et al., 2021).

Displaying the ecosystem accounts as maps were a visual aid in identifying areas and specific habitats for management. In particular, windfarm sites operated under herringbone leases with multiple landowners highlighted that mapping condition scores was useful for informing where on site management decisions for biodiversity could be taken, and which landowners may need stronger encouragement or incentivisation to engage with biodiversity actions. Biodiversity-themed ecosystem accounts can be a useful visual tool to demonstrate how future management can be implemented or improved.

We demonstrate how thematic ecosystem accounts can be developed at the site scale for industry. Ecosystem accounting can facilitate biodiversity decision-making using terminology more familiar to those in industry while also incorporating ecological evidence. The methodology developed here highlights the flexibility of the SEEA-EA framework when compiling accounts, including the ability to select more appropriate ecosystem classification systems that can be mapped with the IUCN Global Typology. We recommend the inclusion of a 'standardisation' step within the SEEA-EA to facilitate the incorporation of complex indicators, especially for biodiversity-themed accounts where indicators can be particularly limited. Delivering accounts as spatial maps highlighted the benefits of this method as a management tool, in addition to forming a baseline of other management tools. Ultimately, small-scale accounting for biodiversity needs to be an iterative co-design process, with involvement and input from ecologists, SEEA-EA experts, stakeholders, and industry end-users. Our accounts were able to highlight the potential for habitat to support biodiversity and identified areas where management may be required to support or enhance biodiversity capacity. While built specifically for windfarms, this work demonstrates a use of the SEEA-EA for site-scale, industry-focused accounts. While elements of developing the accounts were challenging, they did not impede the development of the accounts and adaptations could be made to adjust to specific needs. With increased uptake and usage, barriers to accounting should become less frequent, and site-scale accounting more standardised and accessible.

3.6. Acknowledgements

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Chapter 4 Co-developing biodiversity guidelines for the onshore wind energy industry.

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Author Contributions: EK, JCS, ID and YMB conceived the ideas and designed methodology; EK collected the data; RB conducted the policy analysis; EK analysed the data; EK led the writing of the manuscript. All authors contributed critically to the drafts and gave final approval for publication.

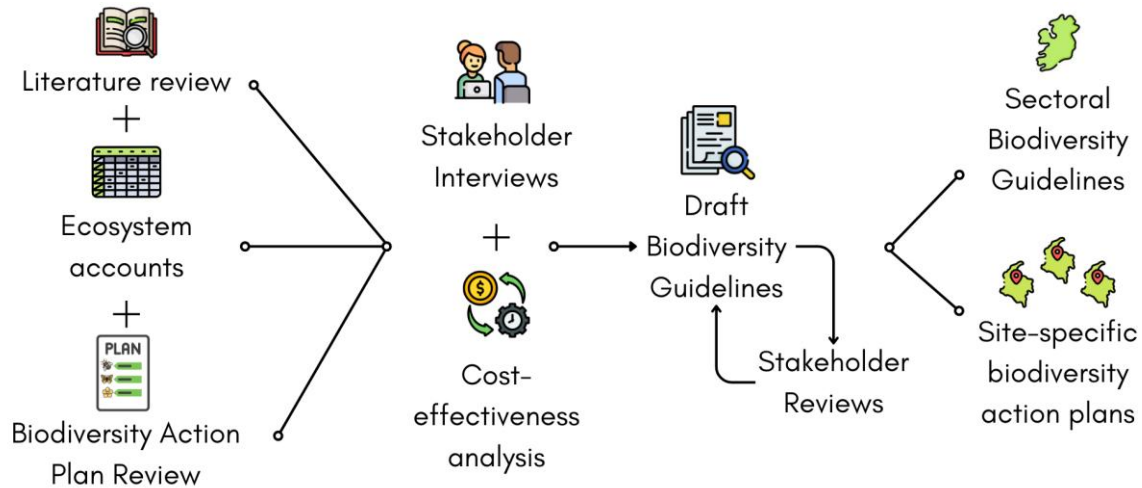
Status: Manuscript in preparation for *Environmental Science and Policy*

4.1 Abstract

Despite recognition of the fundamental role for industry in climate change mitigation and biodiversity protection and restoration, there remain questions about how, why, and for whom, businesses should implement action. The wind energy industry is vital for mitigating climate change and has the potential for substantive positive actions for biodiversity on land surrounding turbines. There is, however, a need for industry-specific guidance to ensure synergistic gains and to prevent or mitigate negative impacts. Although Ireland is a leader in wind energy production, its rapid expansion has led to a gap in the implementation of biodiversity enhancement measures. Using a sample of nine wind farms in Ireland, we compiled biodiversity actions based on substantial existing scientific evidence. Through semi-structured interviews and a cost-effectiveness analysis, we analysed perceptions of industry stakeholders on biodiversity actions. Stakeholders identified several limitations to implementing actions, with cost having a strong effect on perceived cost-effectiveness. Actions such as biodiversity management, habitat creation, restoration, and invasive species management were perceived as highly feasible, beneficial to biodiversity but highly costly. The same action types occurred most frequently in the scientific literature. By aligning the actions with policy, we developed targeted and tailored biodiversity action plans for our sample windfarm sites, and used insights gained to scale these to a set of sectoral biodiversity guidelines. Our findings highlight

the importance of engaging both scientific and stakeholder knowledge in the development of biodiversity guidelines for biodiversity conservation and underscore the importance of collaboration with stakeholders to ensure feasibility and implementation across a sector.

4.1.2 Graphical Abstract



4.2 Introduction

The Earth is facing twin crises of climate change and biodiversity loss, which are intrinsically linked (IPBES, 2019; IPCC, 2023). There is a need, therefore, to implement solutions that address both crises collectively. Biodiversity protection and enhancement can contribute to climate change mitigation and adaptation (Gorman et al., 2023; Shin et al., 2022), and there is growing recognition of the need for solutions to climate change and biodiversity loss to be integrated (Coscieme et al., 2021; Pettorelli et al., 2021). Biodiversity and climate change are increasingly being brought into the private sector (Panwar, 2023; Redford et al., 2015; White et al., 2024), with growing recognition of the potential to provide on-the-ground action for biodiversity in addition to its place in corporate sustainability reporting (Addison et al., 2019). Given the need for transformational change to achieve global biodiversity goals and targets, the role each sector of society can play in contributing to these needs to be explicit (IPBES, 2024; Smith et al., 2020). Collaboration between industry and conservation

scientists, practitioners, local communities, and policy experts is essential to ensure that meaningful positive action is made by businesses (Maxwell et al., 2020), and that co-benefits can be achieved across all aspects of the environment (Shin et al., 2022) and society (de Wit et al., 2025). However, methods for achieving this integration in practice are not well developed.

The ‘Green-Green Dilemma’ is used to describe when a sustainable solution can have consequences for another sustainability element of the system (Dulluri & Raț, 2019). This concept has particular relevance for renewable energy and its potential conflicts with biodiversity conservation (Voigt et al., 2024). One of the largest concerns of this technology are bird and bat vulnerabilities to collision with wind turbines and their blades, which were discovered in early developments in the United States (Kuvlesky et al., 2007; Smallwood & Thelander, 2008). The rapid expansion of onshore wind energy deployment to meet renewable energy targets, and reduce carbon emissions, led to a gap in the understanding of wind energy and wildlife impacts, with both scientific research and policy still needing to catch up to meet wind energy demands and deployment (Gibson et al., 2017; King et al., 2025a (Chapter 2)). This gap has also led to variable interpretations by developers of existing policy and guidelines (Warren et al., 2005), leading to variation in implementation of biodiversity protection, enhancement, and offsets across the industry (Voigt et al., 2024).

For onshore wind energy developments, legal requirements are typically focused on the earliest development stages of windfarms, where the developer is responsible for assessing potential impacts of the new development [e.g., the European Union’s Environmental Impact Assessment Directive (2014/52/EU)], aligning with the mitigation hierarchy (Arnett & May, 2016). Windfarms are increasingly being required to monitor sensitive species during the operational stage as part of their planning consents (IFC, 2023). The rapid growth of the industry and, therefore, rise in the need for biodiversity monitoring has led to a diversity of survey methods for documenting impacts, leading to mixed implementation of monitoring across windfarms, and making comparison of impacts across multiple sites difficult (Conkling et al., 2021). There is a growing body of scientific research identifying monitoring gaps (Davy et al., 2021; Lintott et al., 2016) and highlighting the case for

improved policy and guidelines for biodiversity management at all stages of wind energy (that is, development, construction, operation, decommissioning (Gorman et al., 2023; King et al., 2025a, Chapter 2). Businesses are increasingly recognising and reporting on their commitments to climate change. However, biodiversity disclosures lag behind due to complexity in definitions and indicators (Addison et al., 2019), and a lack of resources and funding compared to climate change (Pettorelli et al., 2021). This is part of the potential explanation for why businesses tend to engage in more symbolic and indirect biodiversity actions, rather than ‘on-the-ground’ conservation, which can be costly and time-consuming (Buxton et al., 2020; Talbot & Boiral, 2021; Wagner, 2023). There is, therefore, a growing need for industry-specific guidance to assist businesses, in particular the wind energy industry, to disclose risk to biodiversity, and choose meaningful and impactful actions for biodiversity conservation.

Existing biodiversity guidelines and conservation plans have been criticized for their lack of scientific underpinning or transparency in sourcing and selecting actions (Downey et al., 2022). To improve success of biodiversity conservation, and promote confidence in the actions taken, there is a need for biodiversity guidelines to be explicitly evidence-based (Sutherland et al., 2004). Frameworks for developing evidence-based conservation plans have been developed in recent years, for example, Priority Threat Management (Carwardine et al., 2012), and the Strategic Evidence Assessment Framework (Sutherland et al., 2021). These frameworks are often aimed at conservation professionals and for large, protected areas. With increasing need to incorporate adaptive management and conservation into a broader diversity of sectors (Redford et al., 2015), there is a need for industry-specific and site-specific guidance also.

Implementing biodiversity actions can be resource-intensive, requiring time, monetary investment, and resources (Buxton et al., 2020). Prioritisation is, therefore, often a key element of biodiversity conservation planning (Game et al., 2013). There has been growing recognition of the need to account for people’s values and integrate stakeholder perceptions into conservation planning (Burbidge et al., 2011; Dean et al., 2024). However, perspectives and motivations for protecting and valuing biodiversity vary both temporally (Mace, 2014) and personally (Admiraal et al., 2017). Stakeholder

engagement therefore plays an important role in understanding values and priorities and ensuring feasibility and implementation of actions. It can lead to new insights that may not otherwise have been identified (Boiral & Heras-Saizarbitoria, 2017), contributing to more beneficial outputs (Jolibert & Wesselink, 2012; Young et al., 2016). For industry, collaboration through engagement is key for establishing trust and ensuring actions are substantive and evidence-based (Dean et al., 2024; Failing et al., 2004). For example, stakeholder engagement has been key in marine spatial planning to support decision-making (Smith et al., 2019). Windfarms can often have similar points of contention to marine spatial planning, with opposing values and views on land use (Ferraro & Ellis, 2017). Stakeholders from the wind energy industry could, therefore, play an important role in identifying industry-specific gaps and knowledge (Hallan & González, 2020). To balance the need for evidence-based actions and prioritisation under finite resources, collaboration between conservation scientists and decision-makers is key to ensuring the feasibility of the conservation plan or guidelines (Game et al., 2013).

The alignment of site specific and sectoral biodiversity actions with regional, national and international policy contexts can provide additional impetus to industry to implement evidence-based biodiversity actions, as they understand how their actions can contribute to societal priorities more generally. Examining this alignment can also reveal which policy documents are more action-based for biodiversity and which ones rely on more generalised or aspirational statements. This highlights areas for policy improvement in terms of a more effective and action-based policy context for biodiversity protection and enhancement through industry and institutional structures working synergistically. In particular, the alignment of site-specific biodiversity actions with regional level goals, targets, objectives and policies (for example, in Ireland's County Development Plans and county-level Biodiversity Action Plans) can help to embed the site-specific biodiversity action plan within the local socio-cultural context, as "landowners, farmers, fishers, foresters and local communities are in many aspects the most important players in biodiversity issues" (NPWS, 2023).

In the Republic of Ireland, deployment of onshore wind energy has shown a steady long-term rise (SEAI, 2024), accelerating in recent years that now delivers the second highest share of wind-generated electricity in Europe (Wind Europe, 2025). This has not been matched with updated relevant Irish guidelines for the sector (González et al., 2016). The existing Wind Energy Development Guidelines date from 2006 with an updated draft produced in 2019, which remains unpublished. Questions on unmet deadlines on publishing the updated version have been raised on multiple occasions in Ireland's national parliament (Dáil Éireann; Oireachtas, 2025a, 2025a, 2025b, 2025c), yet only the 2006 guidelines remain available. Other guidelines that inform Irish wind energy include Local Authority County Development and Biodiversity Action Plans, and species-specific guidance, such as for bats (Bat Conservation Ireland, 2012) and pollinators (National Biodiversity Data Centre, 2021). Apart from the pollinator guidelines, most available guidelines focus on the avoidance of potential impact at the development stage, with little emphasis on enhancement or retaining biodiversity at operational stages. One plausible explanation for the limited sectoral level guidance available could be the challenge of encompassing the intricacies of an entire sector that has a footprint across many habitats and land-uses (Hallan & González, 2020). At the site-level, more recent wind energy developments have provided site-specific habitat management plans or biodiversity guidelines as part of their planning application. Site-level biodiversity guidelines allow for tailored habitat-level guidance, accounting for site-specific nuances. At broader scales, this detail can be lost. Guidance is, however, important at the sectoral scale for highlighting the options available for biodiversity enhancement. There has been a gap in scaling up from site-scale guidelines to sectoral ones. Here, we address that gap by developing site-scale plans that inform the development of sectoral guidelines.

Despite increased recognition of the significant role industries can have in biodiversity conservation (Langhammer et al., 2024), there are still questions about how, why, and for whom, businesses implement action (Boiral & Heras-Saizarbitoria, 2017). Collaboration between industry and other stakeholders plays an important role in addressing such questions and informing conservation decisions (Allison & Arnold, 2018; White et al., 2023a). Here, we outline the co-development of site-specific

biodiversity action plans and sectoral guidelines to explore how collaboration can be used to create scientifically-appropriate guidelines for industry. By using data collected from nine exemplar windfarm sites, we scaled tailored biodiversity action plans into sectoral biodiversity guidelines. Using the wind energy sector as a case study, we demonstrate the scalability and feasibility of using scientific information, policy alignment and stakeholder engagement to develop comprehensive biodiversity guidelines that can be used to facilitate decision-making within an industry that is central to climate change mitigation.

4.3 Materials and Methods

We reviewed the available scientific literature in combination with existing biodiversity action plans to assess the scientific evidence supporting biodiversity conservation actions, aligning this with relevant policy to maximise impact. Given that the compiled actions were to be included in industry-specific guidelines, we engaged with industry stakeholders in semi-structured interviews to gain insights into current knowledge on biodiversity actions. We assessed stakeholder perceptions using a modified cost-effectiveness analysis to understand how stakeholders perceived proposed biodiversity actions. Combining the habitat data, reviews, policy alignment and stakeholder perceptions, we collated site-specific biodiversity action plans and scaled these into a set of biodiversity guidelines for the Irish onshore wind energy industry, which we then presented to the industry for feedback.

4.3.1 Study sites

To represent the variety of conditions and perspectives of a range of industry partners, we selected nine onshore windfarms distributed across Ireland (Fig. 4.1). Sites were selected as representative of onshore windfarms across Ireland, and the habitats and land-uses in which they are typically co-located. Land-use in Ireland is dominated by agricultural grasslands, followed by peatlands, and forests (CSO, 2023), and this was reflected in site selection. Windfarm sites were also selected to represent a mixture of sizes and ages of operation (Table 4.1). Our focal windfarms include upland sites within plantation forestry (3 sites), peatland habitats (2 sites), and

lowland sites located within agricultural grasslands (4 sites). Windfarms ranged in both age (<1 year – 21 years) and size (5 – 36 turbines). The sites were either wholly-owned by the operator or under a herringbone lease, in which ownership is retained by local landowners and the operator leases the area of the windfarm infrastructure. Each windfarm site provided Environmental Impact Assessment Reports (EIARs) and any existing operational ecology reports.

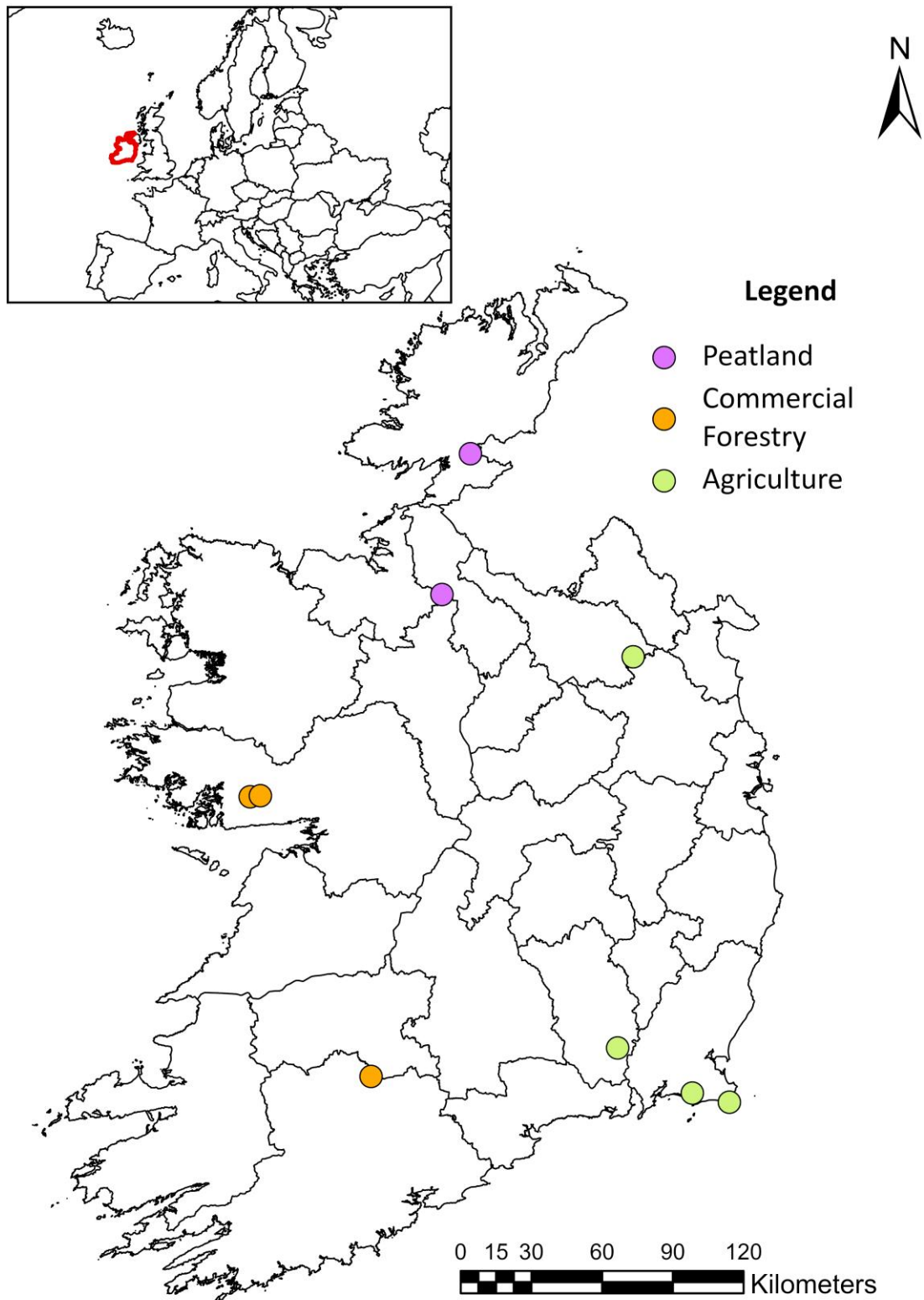


Figure 4.1 Locations of our nine focal windfarm study sites and their dominant land-use. Ireland's location in Europe is shown in red in the inset map.

4.3.2 Local site habitat extent, condition and threats

Site visits were undertaken between June and September in 2022 (seven sites), and 2024 (two sites). A walkover survey of each site was carried out to map habitats, following the guidelines of Smith et al. (2011). Land parcels greater than 400 m² were assigned a code from Level 3 of the Fossitt (2000) *A Guide to Habitats in Ireland* classification system, a standard scheme for assessing habitats in Ireland. Habitats that were transitional and did not fit into one habitat code were given a unique code from combining the two represented habitat types. Codes were assigned to land parcels using ERSI ArcGIS Field Maps software in the field and subsequently digitised in ArcGIS Pro (ESRI, 2023). In addition, data were collected on the condition of the habitats to provide resources for biodiversity, and any risks or threats to that condition observed during site walkover (for details see Chapter 3; King et al., 2025b).

The data collected were transformed into extent and condition accounts using the United Nations System of Environmental Economic Accounting–Ecosystem Accounting (SEEA–EA), as described in Chapter 3 (King et al., 2025b). This provided condition scores for each land parcel, ranging from 0-1, where scores closer to zero indicate the habitat was in poor condition in terms of its capacity to provide resources for biodiversity, and scores closer to one indicate a greater capacity for the habitat to provide resources for biodiversity. These condition scores then could be mapped onto the habitat maps, providing detailed spatial data on areas where more intensive actions for biodiversity may need to be taken, or areas where biodiversity condition is already good and should be maintained.

Table 4.1. Characteristics of our focal study sites for which habitat data were gathered and site-specific guidelines developed. The primary habitat is the most dominant habitat type on the site, defined at the third level of the Fossitt (2000) Guide to Habitats in Ireland.

Land use	Primary habitat	Size (no. turbines)	Year of operation (category)	County
Commercial forestry	Conifer plantation	22	>2021	Galway
	Conifer woodland	36	2011-2020	Galway
	Conifer plantation/Upland blanket bog	6	2011-2020	Cork

Peatland	Upland blanket bog/Cutover bog	14	2000-2010	Roscommon
	Conifer plantation/Upland blanket bog	39	2011-2020	Donegal
Agriculture	Improved agricultural grassland	14	2000-2010	Wexford
	Improved agricultural grassland	5	2000-2010	Kilkenny
	Improved agricultural grassland	18	2000-2010	Wexford
	Improved agricultural grassland	5	2011-2020	Meath

4.3.3 *Compiling and classifying biodiversity actions*

We conducted two complementary reviews. First, we reviewed available literature on biodiversity actions taken at onshore windfarms from 2000-2024. Using Clarivate’s Web of Science database and Google Scholar, we searched for the terms “biodiversity actions”, OR “conservation actions”, with the Boolean operator AND “wind farm(s)”, OR “wind energy”, OR “onshore wind farms”. As searches restricted to Ireland yielded only a small number of studies, the scope of this review was broadened to include global literature.

Second, we reviewed biodiversity action plans developed in Ireland across a range of scales, including the 4th National Biodiversity Action Plan (NBAP) for Ireland (NPWS, 2023). As outlined in the NBAP, all local authorities in Ireland are required to produce a biodiversity action plan by 2026, and smaller scale organisations and local communities are also highly encouraged to develop plans, with grants available to assist this (National Biodiversity Data Centre, 2025). As part of this, the Irish National Biodiversity Data Centre (NBDC) hosts an online platform (*ActionsforBiodiversity.ie*) hosting local and community biodiversity action plans. With this available resource we

were able to search the platform for plans up until August 2024. We also reviewed explicit windfarm guidelines such as the All-Ireland Pollinator Plan *Pollinator-friendly management of wind farms* (National Biodiversity Data Centre, 2021) and the National Wind Energy Development Guidelines (Department of Housing, Local Government and Heritage, 2006). From each action plan reviewed, we extracted each objective and action outlined in the plan. For each action, we indicated whether it had general and broad relevance to the goal of creating a set of actions for the wind energy sector and the feasibility of the action for Irish habitats found on onshore windfarms. For each study site, we also reviewed the relevant County Development Plan, Local Authority Climate Action Plan, and county-level Biodiversity Action Plan (where available), to assess potential synergies and alignment between the regional policy context and the biodiversity actions in the relevant site-specific plan.

We summarised actions from all the biodiversity action plans and guidelines reviewed, grouping them based on similar themes. Once all actions were gathered, we developed nine action types that provided a high-level summary of the goal of the action, such as biodiversity awareness, habitat creation, restoration, monitoring, etc. Each action extracted from the reviewed action plans was assigned to one action type.

4.3.4 Industry stakeholder interviews

Semi-structured interviews were conducted with members of the Irish wind energy industry to develop an understanding of their perspective and knowledge on biodiversity actions for onshore windfarms. An interview guide was developed that included three open-ended questions and a quantitative assessment of the perceived feasibility, biodiversity benefit, and cost of each action arising from our reviews (Table 4.2). We used a likelihood scale from 0 – 100% for both relative feasibility and relative biodiversity benefit, where 0% is impossible and 100% is certain, as used in other stakeholder elicitation exercises (Morgan, 2014; Reyes et al., 2019). To estimate costs, we used a categorical ranking of ‘high’, ‘medium’ or ‘low’ cost, and asked participants to, where possible, estimate the cost of the action (in euros). While interviewees were asked to provide a figure along the likelihood scale, they were also welcome to add additional information to explain their choices, and these were noted by the interviewer. Interviews were held online during 2024 and typically lasted no longer than

one hour. Transcripts of each interview were recorded through Microsoft Teams, and notes were taken by the interviewer throughout. All interviews were completed by the same member of the research team for consistency.

Table 4.2. Questions asked during semi-structured interviews and the answer format.

Question	Answer format
Can you tell me about any actions your company currently takes for biodiversity that you are aware of.	Open-ended
Can you tell me of any biodiversity actions your company is aware of and would like to do in the near future, that you are aware of.	Open-ended
Looking at the list of actions, discuss each action and tell me about your thoughts on what you would perceive their feasibility, cost, and potential biodiversity benefit, to the best of your knowledge, be in relation to conducting the action at a windfarm site.	For <i>feasibility</i> rate on a likelihood scale of 0-100% on how feasible the action would be on a windfarm site. For <i>cost</i> can you indicate whether the cost would be low, medium or high-cost, and where possible estimate what that cost would be in euros. For <i>biodiversity benefit</i> , in your opinion, what is the benefit of the action to biodiversity, on a scale of 0-100%, where 100% indicates it will almost certainly have a beneficial impact, and 0% indicating it's impossible to have an impact.
Can you tell me what the general limitations are of conducting actions for biodiversity are on site.	Open-ended

4.3.5 Cost-effectiveness analysis and guideline development

We conducted a cost-effectiveness analysis using the perceived cost, feasibility and biodiversity benefit industry stakeholders provided during interview. As interviewees provided relative cost as a categorical response variable, this was transformed for analysis into a 1-3 ordinal scale, with low cost converted to 1, medium cost converted to 2, and high cost converted to 3. We performed a multivariate cluster analysis to identify patterns among action types, based on the perceived benefit, cost, and feasibility scores from interviewees. For the cluster analysis, the data were first standardised using z-scores, and then *k*-means clustering was applied. Clusters were

visualised using the ‘ggplot2’ (Wickham, 2016) and ‘ggforce; (v. 0.4.2; Pedersen, 2024) packages in R (version 4.2.2; R Core Team, 2022).

For the cost-effectiveness analysis, we used the raw stakeholder data rather than the standardised z-scores. An indicator of cost-effectiveness (CE) for each action i was calculated as:

$$CE_i = \frac{F_i \times B_i}{C_i}$$

where F corresponds to feasibility (likelihood scale), B to relative benefit (likelihood scale), and C to relative cost (ordinal scale). We calculated the perceived cost-effectiveness of each action per participant and then used the mean cost-effectiveness of the action across all participants to rank actions according to stakeholder perceptions. We also calculated the mean cost-effectiveness score for each of our nine action types.

Informed from the stakeholder perceptions and cost-effectiveness analysis in combination with the habitat condition data for each of the nine windfarm sites we studied, we selected actions and prioritised areas where different biodiversity measures would be suitable to create site-specific biodiversity action plans (BAPs). Suitability was determined based on the site lease type, surrounding habitat, feasibility, and potential biodiversity benefit. We looked at what actions were frequently included in the site-specific BAPs for each land-use type and used this to inform actions that were included in the sectoral guidelines. This plan was not specific to one windfarm site and therefore needed to be generalised across the sector, for a range of habitats, different lease-types, and across all stages of a windfarm’s lifecycle. Actions were tailored to the known constraints of windfarms that were identified during semi-structured interviews, especially for areas under herringbone lease types, where the operator is more limited in the actions that can be implemented. Using the cost-effectiveness analysis and reviews of both actions and impacts, we identified a suite of actions that could be implemented on windfarm sites across Ireland. Common themes and limitations from the stakeholder interviews were included in the plan to further address gaps in industry knowledge.

4.3.6 Industry stakeholder feedback

Following an initial draft of the sectoral biodiversity guidelines, it was important to provide time for members of the wind energy industry to review proposed actions to identify any gaps and address concerns surrounding feasibility or implementation of actions. Feedback was gathered in two forms, through a written review period and a live online review forum. The review process was facilitated through Wind Energy Ireland (WEI), the representative body for the Irish wind industry. The written review was conducted through a draft guidelines document shared with members of the wind energy industry by WEI on 25th November 2024 and reviewers were given until 10th January 2025 to provide initial feedback. Online reviews were conducted by the authors joining meetings of WEI member's working groups, presenting on the guidelines and areas where industry guidance would be particularly important and facilitating discussion on the plan from members. The first online forum was held on 4th October 2024 to the WEI Asset Management Committee for 30 minutes. The second online forum was held on 28th January 2025 for one hour to the WEI Onshore Environment Group.

4.3.7 Data analyses

To understand stakeholder perceptions on collated biodiversity actions we used linear fixed-effects modelling with Weighted Effects Coding (WEC) (te Grotenhuis et al., 2017). We modelled the cost-effectiveness indicator for the nine action types, with interviewee as a random effect, with the aim of testing for differences in perceived cost-effectiveness for different biodiversity action types. Models were built using the 'lme4' package (Bates et al., 2015) and 'wec' package (te Grotenhuis et al., 2017) in R version 4.2.2 (R Core Team, 2022), and visualised using 'ggplot2' (Wickham, 2016). Model fit was checked using the DHARMA package (Hartig, 2022). Linear fixed-effects models were run separately for each component of CE (Cost Effectiveness, that is, perceived cost, benefit, feasibility).

To analyse stakeholder feedback on the draft guidelines, we logged all comments on the document, all email comments, and feedback delivered during the online review forums, and kept a log of how feedback was implemented or why feedback was not implemented in the guidelines. We then used Microsoft CoPilot to

analyse the feedback, specifically asking to code each comment and identify themes in the feedback provided by stakeholders, following the methodology of Turobov et al. (2024). Prompts used to generate results are summarised in Figure 4.2 and the prompt script is included in Appendix C1. We repeated this process 25 times, clearing the CoPilot history after each time to test whether the AI could deliver stable results. We collated the results of each CoPilot replicate run and manually cross-referenced the codes and quotes against the original feedback to check for any errors in the coding/clustering process, in particular known errors such as incorrect quotations and mislabelling (Turobov et al., 2024). We then manually assigned each cluster to a theme. Clusters that varied in terminology between each replicate were simplified into a manual theme, following review of the codes, quotes and cross-referencing with the original feedback which indicated the clusters represented the same codes and feedback (e.g., both “Scientific and technical standards” and “Terminology and communication” were placed under the manual theme “Technical and conceptual clarity”).

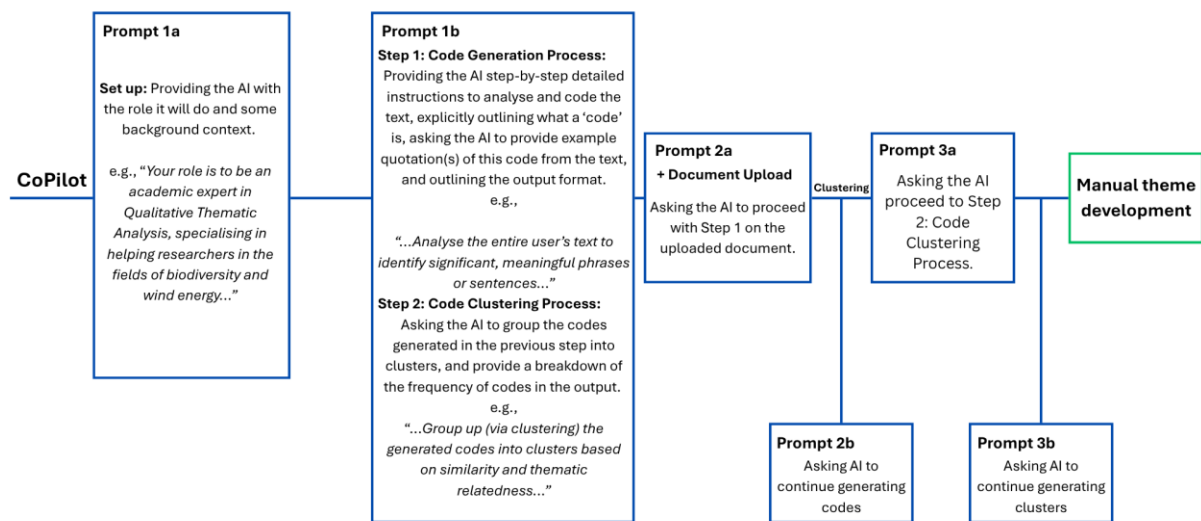


Figure 4.2. Process and summary of prompts followed to code and cluster stakeholder feedback using Microsoft CoPilot, following the workflow of Turobov et al. (2024). Prompts 2b and 3b were used only if CoPilot did not code/cluster the whole document in the first instance. The full prompt script is available in the Supplementary Information.

4.4 Results

4.4.1 Habitat data

There were eight habitat types identified from the wind farms at the broadest level of the Fossitt (2000) *A Guide to Habitats in Ireland* (Level 1), and 34 habitats at the most detailed scale (Level 3). Woodlands and scrub (3973 ha) were the most common habitat type, followed by peatlands (627 Ha), and grassland and marsh (370 Ha). The condition score of the habitats surrounding our focal windfarms ranged from 0.18-0.99, with *Cultivated and built land* (e.g., croplands and turbine hardstands) having the highest average condition score and *Exposed rock* having the lowest condition score (Table 4.3).

Table 4.3. Area and mean condition score (range in parentheses) of habitat types (Fossitt Level 1) across our focal windfarms.

Habitat type	Total area (Ha)	Condition score
Woodland and scrub	3973	0.45 (0.19-0.99)
Peatlands	627	0.50 (0.08-0.87)
Grassland and marsh	370	0.41 (0.18-0.79)
Heath and dense bracken	242	0.50 (0.23-0.79)
Exposed rock and disturbed ground	227	0.37 (0.17-0.67)
Freshwater	73	0.57 (0.2-0.89)
Cultivated and built land	14	0.63 (0.42-0.79)
Littoral	1	0.50 (0.5-0.5)

4.4.2 Reviews of biodiversity actions

We reviewed 33 existing BAPs and recorded 405 biodiversity actions, which we categorised into nine action types. Actions relating to biodiversity management occurred most frequently, followed by raising biodiversity awareness (external) and invasive species (Table 4.4).

Table 4.4. Number of actions per action type gathered from reviewing existing action plans.

Action type	Number of actions
Biodiversity management	127
Biodiversity awareness (external)	87
Invasive species	50
Biodiversity monitoring	30
Biodiversity awareness (internal)	27
Data management	21
Habitat creation	21
Restoration	21

We also reviewed 204 (248 including repeated actions) habitat or species-specific actions across 16 sources (Table C.1). Many these actions came from Conservation Evidence synopses, including Butterfly and Moth Conservation (45 actions), Farmland Conservation (36 actions), Bird Conservation (33 actions), Bat Conservation (25 actions) and Terrestrial Mammal Conservation (21 actions). Actions specific to grasslands had the largest number of supporting actions (121), followed by woodland (35 actions) and peatland (27 actions). Scrub and waterbodies both had the lowest number of supporting actions (2 actions each). For species-specific actions, invertebrates had the greatest number of supported actions (57), followed by birds (34 actions) and bats (32 actions). Vegetation, reptiles and amphibians had the lowest number of supporting actions. From both reviews, we selected 42 actions that, based on habitat surveys and condition scores from site visits, would be appropriate for Irish windfarms and their habitats.

4.4.3 Stakeholder engagement

Nine industry stakeholders took part in semi-structured interviews. In response to actions currently implemented, interviewees frequently mentioned post-construction monitoring for birds, internal staff training, habitat management plans, and pollinator-friendly actions. In response to actions they would like to see implemented, common themes in answers included peatland restoration, bat monitoring, and tree planting. Windfarm stakeholders identified similar limitations to implementing actions on site, which are summarised in Table 4.5, and included resourcing, availability of knowledge, and lease agreements.

Table 4.5. Table of general limitations of conducting biodiversity actions on windfarms identified from semi-structured interviews with windfarm stakeholders.

General limitation	Description	Quote(s)
Resources	Personnel, time, cost. Rising costs can lessen resources for biodiversity actions. Practicality of resourcing actions over the windfarm life-span.	“Time is a limitation on a small team.” “Even small marginal increases in the cost of a project can make it unviable so we have to be aware of that”

Knowledge	Knowing the correct methods, what 'good' for biodiversity looks like and simple dos & don'ts. Don't know where to get knowledge or training.	"Basic knowledge...what are the simple things that we could do, keep an eye out for etc."
Policy/Planning	Environmental Impact Assessment (EIA) is more about protection rather than action/enhancement. More could be done to standardise policy.	"EIA plans are there for the planning and the design, but generally people read EIA plans from the point of view of if we need to protect whatever has been highlighted...and therefore we can't go near that area."
Landowners/Land agreements	Herringbone leases make biodiversity actions more difficult to implement; Buy-in and co-operation is very context and landowner dependent. Difficult to implement if not set out in the lease agreement.	"Limitations [include] consistent landowner engagement, as well as landowner compliance."
Lack of standardisation/guidelines	Using different contractors and having multiple sites makes standardisation difficult, there is currently no standard process for data collection, for example. Lack of guidance on buffers for sensitive species.	"There is no standard guidance on buffer [distances], and this can hold up planning if there are disagreements when it comes to planning."
Lack of engagement with regulators/statutory bodies	Industry needs more engagement and input at the development stage. Often time-consuming and slow return on engagement that delays action.	"If you can't get through the planning system, you won't have any Habitat Management Plans or biodiversity, and the gain associated. I think, engagement with regulators and statutory bodies would be very useful to have."
Planning	Delays in planning meant that ecological reports get outdated. Operators aren't incentivised to do actions if not part of planning.	"Things take a really long time to get through...we've had projects stuck in planning for seven years...It's just not doable now."
Data	Consultant vs company access to data can differ. Difficult to make decisions if don't know what data are there, especially with	"If you have 36 sites and you've slightly different reports on each of them...you could have a lot

movement of personnel. Collected data are underutilised.	of data management-type issues.”
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4.4.4 Cost-effectiveness analysis

Clustering analysis grouped the nine biodiversity action types into three clusters based on the mean relative benefit, relative feasibility, and cost provided by interviewees (Fig. 4.3). The first cluster included action types perceived to be high benefit to biodiversity, highly feasible at onshore windfarms in Ireland, and low cost, including action types: data management, and raising awareness about biodiversity. The second cluster included action types that were perceived to be of high benefit and high feasibility, but with higher cost, and included biodiversity monitoring, invasive species management, and restoration actions. The third cluster comprised only one action type, citizen science, which was perceived to be of lower benefit and feasibility, and low cost.

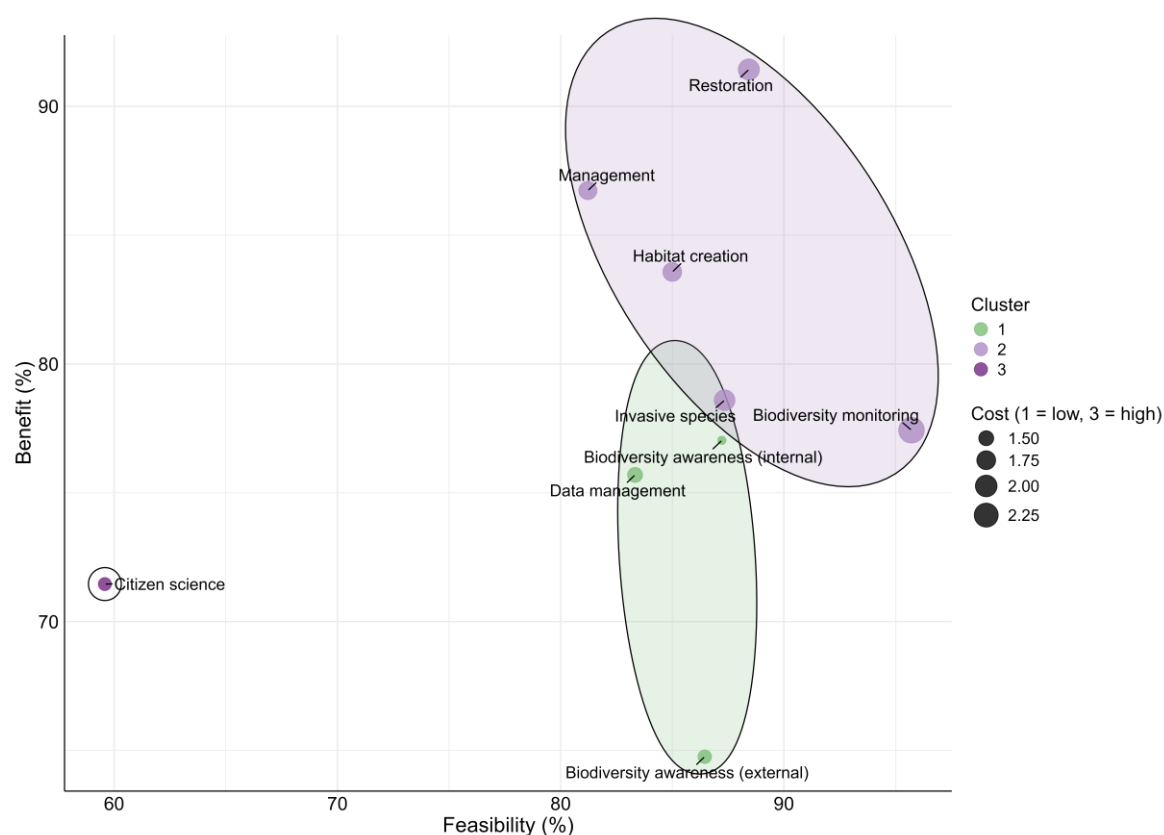


Figure 4.3. The mean feasibility and relative benefit of the nine action types across interviewees, with relative cost shown as point size. Cluster 1 (green) comprises actions that were perceived to be highly feasible but with lower benefit and lower cost, Cluster 2 (light

purple) comprises actions that were perceived to be highly feasible, with higher benefits and higher costs, and Cluster 3 comprises a single action with lower feasibility and benefit together with a low cost.

We found a significant and positive effect of both biodiversity awareness (internal) and data management on perceived cost-effectiveness, and a negative effect of *biodiversity monitoring* and *citizen science* (Fig. 4.4a, Table C.2). Feasibility of an action varied across the action types, with *biodiversity monitoring* identified as the most feasible action type while *citizen science* was the least feasible (Fig. 4.4b, Table C.3). Perceived benefit of an action also varied across the action types, with *restoration* and *management* having the highest estimated effects, meaning they are associated with the greatest increase in perceived benefits and *raising biodiversity awareness (externally)* having the lowest perceived benefit to biodiversity (Fig. 4.4c, Table C.4). *Biodiversity awareness (internal)* and *citizen science* had the largest effect size for cost, indicating interviewees perceived these to be lower cost action types, while *biodiversity monitoring* had a significant negative impact on effect size, indicating interviewees perceived this action to be associated with high costs (Fig. 4.4d, Table C.5).

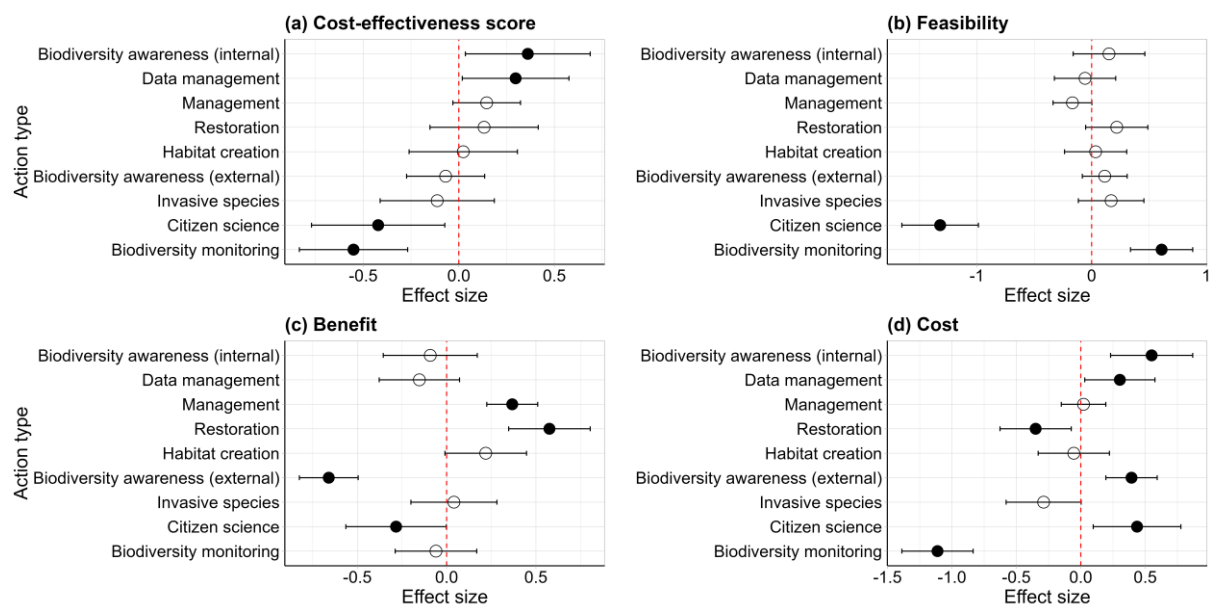


Figure 4.4. Effect sizes for each of our nine action types for cost-effectiveness (a), feasibility (b), benefit (c) and cost (d). All plots display the results of linear mixed-effects models based on scores elicited during semi-structured interviews with members of the wind energy industry. The y-axis of each plot follows the order of plot (a) Cost-effectiveness score, and highlights differences in preferred actions across cost-effectiveness and its various

components, with higher effect size indicating preferred actions, and lower effect sizes indicating least preferred actions. To align visually with the other plots (a, b, c) costs (d) were re-coded to 1 (high cost) and 3 (low cost) so that effect size followed the interpretation of the three preceding plots. Error bars represent 95% confidence intervals, statistical significance ($P < 0.05$) is indicated by closed points, with non-significant effects shown with open points.

4.4.5 Industry stakeholder feedback

Overall, we received 81 pieces of feedback from industry, from comments on the draft document (44), by email (21), and through the online forum (16). CoPilot produced 108 thematic clusters across the 25 replicates we conducted of the analysis in the AI software, many of which were very similar (e.g., *Policy and Governance* and *Policy and Governance Gaps*). CoPilot produced an average of 7 clusters per replicate of the analysis (range 4 – 10). The manual theme development process resulted in 8 themes (Table 4.6). The clusters produced by Microsoft CoPilot and the manual theme they were grouped into are described in Table C6.

Additional actions from the 42 originally presented to stakeholders during the semi-structured interviews arose from stakeholder feedback and recommendations. The resulting plan included 56 specific actions across eight action types (Table C7). The full sectoral guidelines can be found at King et al., 2026 and an anonymised example site-specific biodiversity action plan can be found in Appendix C6.

Table 4.6. Themes assigned during the manual theme development process with exemplifying quotes from the feedback from stakeholders.

Theme	Exemplifying quote
Biodiversity-Planning Integration	“Could we look at the corresponding policy in the UK that may become common practice in Ireland over time, such as Biodiversity Net Gain. Can Biodiversity Net Gain be added to the Biodiversity Action Plan, in terms of for example the lessons learnt from application in the UK or possible methods of application in Ireland.”
Climate-Biodiversity Nexus	“Possibly some reference could be given to the urgency of the climate crisis in the Green-Green Dilemma and the impact of climate change on biodiversity (both habitats and species). Wind developments can take an average of 10 years from development to operation (some up to 20 years). Immediate action is required if we are to meet our climate

	change targets by the end of the decade and avoid significant penalties (and further waste of limited resources). The adage of 'Right action, Right place' is correct, but unfortunately actions also need to be 'Right now' which adds to the challenge of delivering for climate and biodiversity targets.”
Landowner Engagement and Lease Agreements	“There are many different options for lease agreements in terms of biodiversity, these should be considered.”
Monitoring and Data Requirements	“Who would be responsible for providing a central database for biodiversity data to be stored and easily accessed?” “We also monitor the actions, or should, implemented to mitigate effects for efficacy and change or adapt these actions where needed.”
Planning, Development & Operational Constraints	“Planning requirements for restoration work (e.g., drain blocking and conifer removal on peatlands) may become an issue depending on the scale and type of peatland restoration, and we have seen some evidence of this from local authorities. Further planning requirements for restoration works could lead to significant delays and could prohibit some restoration works. Highlighting this may be useful for highlighting potential issues and possibly finding an early solution.”
Policy and Governance Gaps	“Are there any recommendations for a national approach or policy of assessing progress towards these goals over the lifetime of the development (c.30 years) and could biodiversity net gain on wind developments contribute to national targets under an appropriate mechanism?”
Restoration and Habitat Management	“Distinguishing between peatland types for restoration as they have different methods.” “If developers are considering rewilding hardstands, it would be useful to note possible operational risks around functionality and safety so locations can be assessed on a site-by-site basis.”
Technical and Conceptual Clarity	“Clarity is needed around terminology and definitions.” “Are there any issues around felling licences with this measure [replanting with native woodland]? They are prescriptive on replanting requirements.”

4.5 Discussion

Using a multifaceted approach to combine scientific and stakeholder knowledge, we developed a set of biodiversity guidelines to support the onshore wind energy industry in Ireland to take positive actions to support biodiversity. We found substantial existing support and evidence available for biodiversity-positive actions. Through a semi-structured interview process combined with a cost-effectiveness analysis, we identified gaps and concerns in stakeholder perceptions of biodiversity actions. Clustering of stakeholder perceived cost-effectiveness identified the most scientifically-evidenced actions as being highly feasible and beneficial to biodiversity, but of high cost. Cost-effectiveness indicators highlighted stakeholder preferences for biodiversity actions, with raising awareness about biodiversity within staff and data management ranked the highest for perceived cost-effectiveness, while biodiversity monitoring and citizen science were perceived to be the least cost-effective action types. Stakeholder feedback played an important role in highlighting the need for improved communication between scientists and stakeholders. Engagement with stakeholders and alignment with available policy highlighted the importance of the role policy can have in incentivising and supporting uptake of biodiversity actions and furthering the need for biodiversity guidelines to be aligned with policy. The methods applied here provided an efficient way of developing a tailored set of biodiversity guidelines at two scales (site-specific and national) that are supported by scientific evidence and aligned with policy, which can be used to support biodiversity in a sector that is vital for climate change mitigation.

With growing awareness of the biodiversity crisis, there is increasing pressure on industries to disclose and implement actions they are taking for biodiversity (Ali et al., 2024). However, companies may lack the internal expertise to deliver meaningful impact (e.g., from stakeholder interviews *“Time, and particularly staff time and resources are limited, which makes actions more difficult and expensive to implement”*). Thus, collaboration between industries that want to take positive action for biodiversity and conservation scientists can result in realised successful action on the ground for biodiversity (Blicharska et al., 2016; Gallo et al., 2018a). In addition, stakeholder-scientist collaboration draws on different perspectives and knowledge

(Addison et al., 2019), and identifies priorities, values, limitations, and potential conflicts (Admiraal et al., 2017; MacLeod et al., 2022). By engaging stakeholders early, these learnings can be explicitly addressed to ensure tailored, context-specific knowledge is generated and applied (Norström et al., 2020). Although from varying roles across the wind energy sector (e.g., project manager, asset manager, planning and environmental officer), each stakeholder interviewed in this study had knowledge of the current actions their companies were taking for biodiversity and could also provide examples of actions they would like to see implemented on their windfarm sites. This provided important industry-specific context as to what actions are currently used on windfarms, as this information is not usually published or can only be found in grey literature. The current biodiversity actions used by windfarms also highlighted the uptake of existing guidelines and initiatives, such as the All-Ireland Pollinator Plan ‘Pollinator-friendly management of windfarms’ (National Biodiversity Data Centre, 2021) and the Hare’s Corner initiative (BurrenBeo Trust, 2025).

We identified key limitations to implementing biodiversity actions at onshore windfarms in Ireland. These included a lack of resources (personnel, time, costs, etc.), knowledge availability, gaps in policy and planning regulations, landowners and lease agreements, and data availability and standards. These limitations could also be seen in responses to the cost-effectiveness and clustering of actions. For example, monitoring costs were frequently raised as a limitation to implementing biodiversity actions (e.g., “*Monitoring costs, especially justifying it when it can have little benefit to biodiversity*”), and this was seen when biodiversity monitoring was seen as costly, and the least cost-effective in the Cost Effectiveness and cost models. Limitations such as lack of resources and knowledge gaps arose both in stakeholder interviews and feedback, and are frequently found in biodiversity conservation (Hermoso et al., 2022; Hoffmann, 2022; Ní Shúilleabháin et al., 2023). Some limitations that arose are specific to the Irish context in which this study is placed. For example, comments on gaps in planning regulations and slowness in the planning system relate to previously identified pitfalls in implementing Irish wind energy deployment (González & Connell, 2022; Hallan & González, 2020). Knowledge and expertise are also more generally mentioned as limitations of implementing conservation actions, and this relates to the

complexity of understanding ecosystems (Maxwell et al., 2020), the accessibility of scientific knowledge (Downey et al., 2022) and the lack of capacity within the ecological sector to match demand (CIEEM, 2022). With the purpose of developing guidelines in mind, gathering stakeholder perceptions on limitations facilitated their incorporation into the guidelines in terms of alternatives or considerations to work around the limitations. For example, stakeholders were sceptical of the feasibility and benefit of citizen science, thus it was reworked in the guidelines with the raising awareness about biodiversity action and included as actions such as “Community/Visitor iNaturalist identifications”. This highlighted the dual benefits of data collation and community engagement. Moreover, feedback critiqued an action on letting hardstands rewild for pollinators due to health and safety concerns, therefore this action was amended to “Create a pollination zone around the turbine hardstand” which involves a one metre buffer adjacent to the hardstand and more biodiversity-friendly management of the hardstand itself, while maintaining health and safety protection.

Cost-effectiveness of conservation actions is an important consideration for industries taking action (White et al., 2023b). Asking stakeholders to rank each component (cost, feasibility, and effectiveness) allowed us to understand stakeholder priorities in terms of implementing actions. Stakeholders often struggled initially with cost, even when using a categorical ranking of low-, medium-, or high-cost, and few provided an estimate of the realised cost of the action. Costing conservation actions has been recognised as a challenge for developing conservation cost-effectiveness (Pienkowski et al., 2021), and there have been calls to standardise reporting of conservation costs to overcome this limitation (Iacona et al., 2018). Presenting realised costs to stakeholders would improve the outcomes of the cost-effectiveness analysis. It is, therefore, recommended that, for future guideline development, realised costs are collated prior to the stakeholder engagement phase.

Analysing the components of the cost-effectiveness indicator separately helped explain the results of the cost-effectiveness analysis. For example, biodiversity monitoring is a planning requirement at the development and early operational stages (Wang & Wang, 2015), so stakeholders provided examples of their reasoning for their

answers when ranking this action. This is evident in that biodiversity monitoring was perceived as highly feasible and costly, which aligns with findings from a review of species management plans that species with higher costs allocated to monitoring had poorer outcomes (Buxton et al., 2020). This indicated that, within the guidelines, emphasis was needed on how the outputs of biodiversity monitoring could be optimised to benefit biodiversity. Windfarm monitoring is often driven primarily by policy and regulations, rather than scientific evidence (Copping et al., 2020). Stakeholders could see the value in improving the use of monitoring data, with the action type '*Data management*' being identified as one of the most cost-effective action types. This action type included actions such as '*Making use of open biodiversity data to inform decision-making*' and '*Providing a central database for biodiversity data to be used and easily accessed*'. These findings are consistent with those of González & Connell (2022), which found that centralising data access would be an important step in improving the siting of Irish renewable energy developments.

Citizen science scored low across all components except cost. Stakeholders noted accessibility to windfarms as a potential limitation, in addition to questioning the use of the data collected to make informed decisions. Citizen science has, however, been used successfully on windfarms to inform biodiversity conservation (Newson et al., 2017). In the biodiversity guidelines, we incorporated actions that involved citizen science into the actions on raising awareness about biodiversity as this may encourage stakeholders to implement the actions, as it focuses more on the engagement and empowerment of local communities to assist biodiversity, rather than on data collection. Monitoring and managing invasive species were ranked as costly and stakeholders had a range of responses on actions relating to the biodiversity benefit of monitoring, avoiding and removing invasive species. This result possibly relates to the idea that even for feasible invasive species actions such as mapping and monitoring the location of invasive species, it would still require expert ecological knowledge. Given that invasive species control and management have been shown to be some of the most effective conservation actions (Langhammer et al., 2024), there is a knowledge gap and therefore emphasis was placed in the guidelines on the conservation value of conducting such actions.

There were several common themes in the feedback from stakeholders on the draft guidelines, some relating to more technical aspects, such as clarification of terms and adding definitions to the text, and others which related to the potential pitfalls or limitations of implementing the biodiversity actions in practice. Researchers contribute important scientific knowledge but do not sit in on the day-to-day operations of the windfarm, which means potential conflicts of a biodiversity action with windfarm operations may be missed. Thus, the feedback provided an important communication pathway that ensured any potential issues could be addressed within the document early on in its development. Some of the feedback correlated with the limitations identified during the semi-structured interviews and identified where we should strengthen the arguments and devise workarounds in the guidelines. For example, feedback relating to policy and governance gaps were similar to what was elicited as a limitation during the semi-structured interviews. Given that environmental policy and renewable energy policy are rapidly changing areas (Gaffney et al., 2017; Hering et al., 2023), it can be difficult to be specific and explicit when addressing these in the guidelines. However, highlighting the alignment of specific biodiversity actions with regional, national, and international policy contexts underlined where the policy context supports industry in biodiversity conservation and enhancement. In particular, making industry aware of the policy alignment of relevant biodiversity actions helped demonstrate that the actions were underpinned by prevailing policy as well as scientific research. This has the potential to motivate industry to be ‘ahead of the game’ by implementing such biodiversity actions even if not (yet) legally required to do so.

By building a knowledge base from exemplar wind farms through application of scientifically collated actions to specific habitats and their condition for biodiversity, we gained valuable insights that informed and aided scaling actions into a sectoral-level set of guidelines. Scaling from site-scale to broader scales can lead to tensions (Wyborn & Bixler, 2013), with the biggest problem in scaling up to sectoral levels being the loss of specificity and detail that could be included. Given that the intention of the sectoral guidelines was to support the industry at all onshore windfarms in Ireland, it had to be kept to a broad detail and account for the many different types of habitats

and land-uses onshore windfarms are located within. We included caveats and listed the necessary conditions for each action that was included. The insights gained from developing the site-level biodiversity action plans and using them to inform the sectoral guidelines was greatly beneficial to understanding the intricacies of the specific sector, which may have otherwise been missed. The resulting guidelines provided a general overview of potential actions that can be applied to onshore windfarms in Ireland, accounting for the windfarm's life-stage and the land-use/habitats which it is sited within.

There is growing recognition of the role industries can play in mitigating the twin crises of climate change and biodiversity. However, industry-inclusive, scientifically-supported, and policy-aligned tools to support implementation of action are lacking. By using a multifaceted approach, we applied knowledge gained from multiple sources to develop tailored and site-specific biodiversity action plans, which we then scaled to biodiversity guidelines for the sector. By including stakeholders at several steps in the process, industry knowledge was included in the design and development of the guidelines, which should support their uptake and use in decision-making (Norström et al., 2020). Asking stakeholders both for input (through semi-structured interviews) and feedback builds trust in the process and this is a key element for conservation success (Young et al., 2016). It allows the guidelines to address nuances that may otherwise be missed and allows for opportunities that may not otherwise arise under traditional segregated methods (MacLeod et al., 2022). In addition, the knowledge gained from site-specific action plans allowed us to adapt actions to the land-uses and habitats windfarms are typically sited in within the Irish context, which provided us with context for the literature reviews and policy alignment that we could implement into our plans and guidance. Although the approach taken here was used to develop action plans and guidelines for the Irish onshore wind energy sector, the methodology used could be adapted to any industry that has impacts on the ground and requires biodiversity guidance. Industry-specific guidance and implementation of biodiversity action can facilitate synergistic gains for both climate and biodiversity across sectors.

4.6 Ethics Statement

Ethical approval for the semi-structured interviews was granted by the Trinity College Dublin Data Protection Office.

4.7 Declaration of generative AI and AI-assisted technologies

During the preparation of this work the authors used Microsoft CoPilot to identify themes within industry feedback. After using this tool, the authors manually cross-referenced the outputs and reviewed and edited the content as needed and take full responsibility for the content of the published article.

4.8 Acknowledgements

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Chapter 5 Bat activity at Irish windfarms

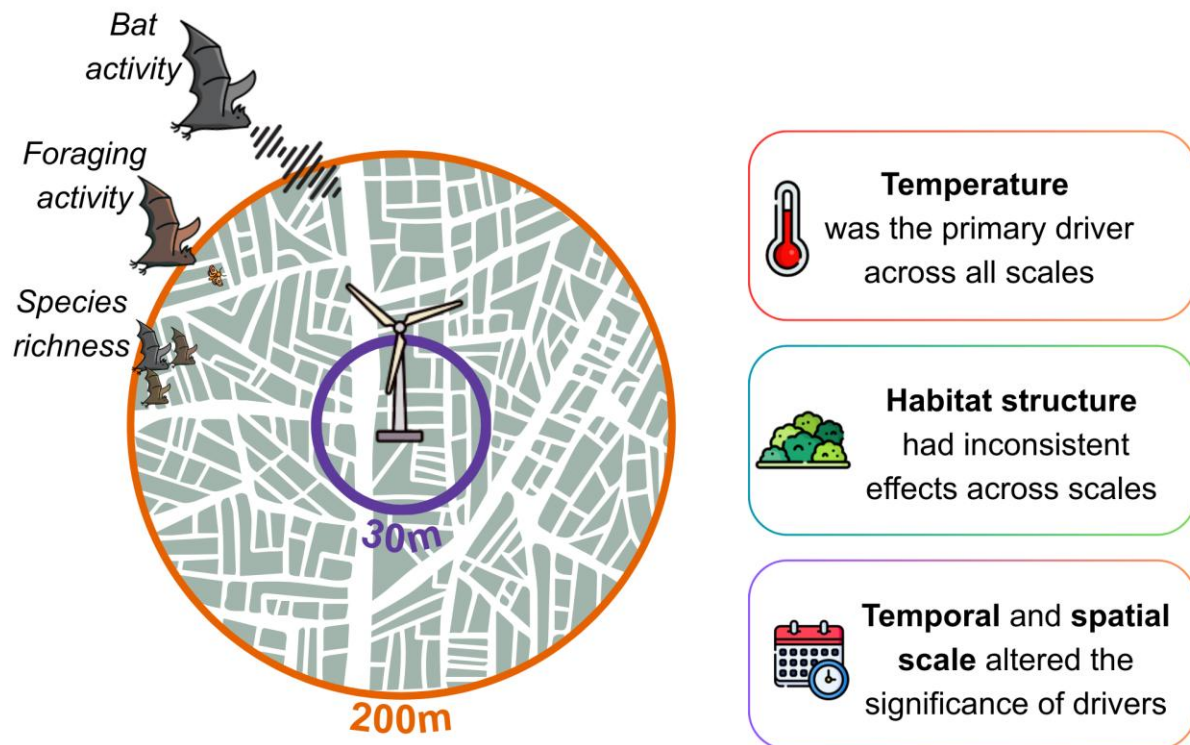
Authors: Emma King, Yvonne M. Buckley, Jane C. Stout, Ian Donohue

Author Contributions: EK, YMB, JCS and ID conceived the ideas and designed methodology; EK collected the data; EK analysed the data; EK and ID led the writing of the manuscript. All authors contributed critically to the drafts and gave final approval for publication.

5.1 Abstract

Bats are vulnerable to renewable energy expansion through collision risk and habitat loss. Understanding bat activity near operational wind turbines and the drivers of this activity across spatiotemporal scales is crucial for effective mitigation. Here, we show that bats actively use areas near turbines in agricultural landscapes and that habitat effects depend critically on the scale of measurement. Using high resolution passive acoustic monitoring at three operational windfarms in agricultural landscapes in Ireland, we analysed how local habitat structure at fine (30 metre), and landscape (200 metre) scales influenced bat activity, foraging activity, and species richness across sub-hourly, nightly, and weekly temporal scales. Temperature was the strongest and most consistent driver, with increased bat activity, foraging, and species richness associated consistently with warmer conditions across all spatiotemporal scales examined. Habitat structure (habitat diversity and linear feature density) showed scale-dependent effects, influencing species richness at fine spatial resolutions but not at the 200 m scale recommended by EUROBATs siting guidelines. Current guidelines for siting wind turbines to protect bats may therefore miss important fine-scale habitat drivers. We recommend multi-scale, context-dependent approaches to windfarm assessment rather than standardized buffer-based metrics.

5.1.2 Graphical Abstract



5.2 Introduction

Bats are attributed protected status in many regions globally [e.g., EU Habitats Directive Annex IV (Council Directive 92/43/EEC, 1992)], and contribute to a range of economically important ecosystem services, such as pest control (Boyles et al., 2011; Kunz et al., 2007). Even so, over one third of bat species are threatened globally (Frick et al., 2020). Bats are sensitive to many threats, including climate change (Festa et al., 2023; McGowan et al., 2021; Rebelo et al., 2010), disease (Hoyt et al., 2021; Mühldorfer et al., 2011), human conflict (O'Shea et al., 2016), land use change and habitat fragmentation (Browning et al., 2021; Mathews et al., 2015; Treitler et al., 2016). A major driver of land use change affecting bats is the need to increase reliance on renewable energy to reduce carbon emissions to help alleviate climate change (Poggi et al., 2018). Wind energy is one of the most rapidly deployed renewable energy sources (European Commission Directorate-General for Energy, 2025).

Research into the impacts of windfarms on bats began to expand rapidly following the discovery of collision risk (Johnson et al., 2003), especially in the USA

(Kunz et al., 2007) and more recently in Europe (Gaultier et al., 2020). Impacts and fatalities are particularly troubling for bat species due to their slow life histories as long-lived mammals that are slow to reproduce, making their populations particularly vulnerable to declines (Baerwald et al., 2009; Barclay et al., 2003). Furthermore, collision with wind turbines is just one potential impact of wind energy, and additional impacts include habitat loss and fragmentation from turbine siting and construction (Carlier et al., 2019; Ellerbrok et al., 2024), disturbance due to turbine noise (Schaub et al., 2008), displacement from commuting routes or foraging locations (Reusch et al., 2022; Scholz et al., 2025b), and barotrauma (Baerwald et al., 2008; although see Lawson et al., 2020 and Rollins et al., 2012). The cumulative impact of these effects is relatively unknown, and is likely to be site-specific (Roscioni et al., 2014).

The site-specificity of windfarm impacts, combined with the inconspicuous activity of bats due to their ecology, adds to the challenge of understanding cumulative impacts on bats (Leroux et al., 2023). Collision risk has been correlated with favourable weather conditions for bat activity (e.g., warmer temperatures, low rain and low wind; Cryan et al., 2014; Rydell et al., 2010). Furthermore, collision risk is also dependent on the landscape context in which the windfarms are sited—collision risk is, for example, lower in flat open farmland compared to on forested hills and ridges (Rydell et al., 2010). Impacts of wind turbines can also be highly species-specific, as species have different habitat preferences and tolerances, and so some are more adaptable than others (Scholz et al., 2025a). Behaviour of some bat species has been found to vary in the proximity of wind turbines (Barré et al., 2018; Millon et al., 2015), with some species even found to utilise turbines as foraging sites (Horn et al., 2008). Proposed explanations for this include attraction to insects that swarm at turbines (Cryan & Barclay, 2009), or mistaking turbines as trees (Cryan et al., 2014). There is, therefore, a clear need to better understand how both habitat and species-specific characteristics impact bat activity at windfarms.

Given their nocturnal lifestyles, bat activity can be relatively difficult to measure (Russo et al., 2021). The development of passive acoustic monitoring presents an important tool to measure the activity of an otherwise difficult to study group (Jones et al., 2009) and has facilitated the development of our understanding of bat behaviour.

Recordings of bat audio contain an abundance of ecological information that can inform about bat activity and behaviour (Prat et al., 2016). Acoustic sampling can, therefore, be a powerful tool to further understanding of how bats utilise windfarms. There is, however, a need to understand its limitations prior to implementation (Skalak et al., 2012). The minimum recording duration required to sample bats reliably depends on the sampling effort (e.g., number of recorders), weather conditions, species abundance, and species behaviour and vocalisations (Richardson et al., 2019; Roemer et al., 2025; Skalak et al., 2012). In a study assessing the statistical power of acoustic surveys, Fischer et al. (2009) found evidence supporting the use of multiple detectors within a site but also found that other environmental aspects, such as seasonality and insect availability, affect activity levels. Further technological advances have led to automation of bat species identification through artificial intelligence-based modelling tools, improving the efficiency and broadening the availability of acoustic data collection (Rydell et al., 2010). In addition to informing bat species presence and activity rate, acoustic monitoring combined with weather patterns can be used to inform predictive algorithms for turbine curtailment as a mitigation tool for collisions (Behr et al., 2017).

Despite the need to understand how bats are utilising windfarms as a feeding resource, few studies have included foraging in their analyses of bat activity near windfarms (Millon et al., 2015). When foraging, bats emit ultrasonic pulses. Once a bat finds prey the interval between pulses then decreases, creating a feeding buzz (Fenton, 2003). These calls can be recognised by visual inspection, but more recently this process can be automated (Jameson, 2024). There is, therefore, a case for utilising the structure of bat calls to understand bat activity and foraging activity at wind turbines and, importantly, to test what influences these activity measures at different scales (Gallo et al., 2018b) to understand how bats utilise areas around wind turbines.

Nine bat species are found in Ireland, all of which are protected under the European Union's Habitats Directive (Council Directive 92/43/EEC, 1992) and the Wildlife Act (1976). Indeed, Ireland is a stronghold for a number of bat species (Marnell et al., 2019). Furthermore, Ireland is a member of The European Union (EU) Bat Agreement, EUROBATs, which provides protection for bat species as part of the

Convention on the Conservation of Migratory Species of Wild Animals (1979), including guidelines for wind energy operators regarding bats (Rodrigues et al., 2015). While there has been great progress in recent years on expanding knowledge of Ireland's bat species and their activity (Pickett et al., 2019; Roche et al., 2007, 2025), there is still limited understanding on how bats are impacted by wind turbines in Irish landscapes. Much knowledge is instead inferred from other regions with similar landscape contexts, such as the threat of windfarms to Nathusius' pipistrelle (*Pipistrellus nathusii*) (NPWS, 2025). It is also not understood how the impacts of windfarms will be confounded with other impacts facing Irish bat populations, including habitat loss, fragmentation and other forms of land use change (Carlier et al., 2019). Further, it is also poorly understood whether bats are migrating between Ireland and the United Kingdom and mainland Europe (Hooker et al., 2025) and it has been established that wind energy can impact migration patterns (Frick et al., 2017; Voigt et al., 2016). Additionally, it has been postulated that bat species richness will increase in Ireland under future climate change, with ranges of many species shifting northwards (Festa et al., 2023; McGowan et al., 2021). Indeed, evidence of this is already seen with the range expansion of Nathusius' pipistrelle (Lundy et al., 2010). Thus, it is a critical time to further understand how Irish bat species are impacted, as in the future we may also need to account for northward shifts of species not currently in Ireland (Fialas et al., 2025).

In Ireland, bats are typically deemed to be of low risk for collision with turbines, and until recently were not specifically assessed in the planning stages of windfarm developments (Chapter 2; King et al., 2025a). Even when they are included in surveys during the planning process, this may not be sufficient to identify threats (Froidevaux et al., 2025). Even if collision risk is low, there could be knock on non-collision impacts for bats (e.g., displacement, habitat loss, fragmentation of habitats), and we currently do not know whether and how bats are using land in the vicinity of windfarms. Many of the first windfarms in Ireland were set in areas now understood to be of higher collision risk for bats, such as forested hillsides (González et al., 2016; Rydell et al., 2010). As renewable energy demand increases, more planned developments are now being co-located within agricultural land use settings (González et al., 2016; Hallan & González,

2020). Thus, there is a need to understand whether and how bats utilise turbines in areas that are currently considered as low-risk and, if they are using these habitats, what influences this activity.

Here, we explore this by addressing the following two research questions: (1) what is the relationship between local habitat structure (measured as habitat diversity and linear feature density) in the proximity of onshore wind turbines and the overall activity, foraging and species richness of bats? and (2) do these relationships vary across temporal (that is, 10-minute windows vs nightly vs weekly) or spatial (that is, 30 metre vs 200 metre radii around recorders) scales? We hypothesise that bat activity, foraging activity and species richness will be driven by habitat structure surrounding wind turbines, and that these drivers become particularly evident at the coarser temporal (that is, nightly and weekly) and spatial (200 m) scales.

5.3 Materials & Methods

5.3.1 Study area

The study took place on three onshore windfarm sites located in the east of the Republic of Ireland. These sites were selected because of their similarity in land use and surroundings—the windfarms were located within an agricultural landscape, primarily pastoral farming within the immediate surroundings of the windfarm infrastructure, and small areas of tillage and commercial forestry. The study sites varied in the duration of windfarm operation and the type and number of installed turbines (Table 5.1).

Table 5.1. Windfarm characteristics and weather station details for each study site.

Site Name	County	Site Area (Ha)	Initial Year of Operation	Number of Turbines	Turbine Model	Weather Station	Weather Station Location
Carnsore	Wexford	79	2002	14	Vestas V52/850	Johnstown Castle	52.2939, -6.5012

Richfield	Wexford	126	2006	18	GE Energy	Johnstown Castle	52.2939, -6.5012
Teevurcher	Meath	64	2018	5	Senvion MM82/2050	Ballyhaise (Hourly data) Carrickmacross (Daily data)	54.0516, -7.312 53.9781, -6.7183

5.3.2 Bat call recording

Bat calls were recorded from 28 June – 21 August 2024 using Wildlife Acoustics Passive Acoustic Song Meter Mini recorders (SMMINI2-AA). For each of the five wind turbines per site, one recorder was placed on a fence post or suitable post approximately 1-2 metres above the ground within 30 metres of the turbine. Recorder Wind turbine locations were selected to minimise overlap of 200 metre buffers around the wind turbines (Fig. D.1, D.2, D.3), aligning with EUROBATS guidelines for distance from woody features (Rodrigues et al., 2015). In some locations, due to the nature of the site, it was not possible to avoid some overlap. Recorders were set to record each night, from 30 minutes before sunset until 30 minutes after sunrise each day.

5.3.3 Data sources

We used habitat maps from our field-based habitat surveys conducted in 2022 (Chapter 3; King et al., 2025b), followed by site walkovers in 2024. If any deviations were found, the difference was noted on a field map, and the digitised map was updated. The maps were developed from a walkover survey of each site, following the guidelines of Smith et al. (2011). Land parcels greater than 400 m² were assigned a code from Level 3 of the Fossitt (2000) *A Guide to Habitats in Ireland* classification system, a standard scheme for assessing habitats in Ireland. Habitats that were transitional and did not fit into one habitat code were given a unique code from combining the two represented habitat types. Codes were assigned to land parcels

using ESRI ArcGIS Field Maps software in the field and subsequently digitised in ArcGIS Pro (ESRI, 2023). Linear features, such as hedgerows, treelines, and earth banks were also recorded in the field and digitised. Habitat data were clipped to 30 m and 200 m buffers around each recording location, and the area of each habitat type was calculated in ArcGIS Pro (ESRI, 2023). Simpson's Diversity Index (Simpson, 1949) was calculated from the area of each habitat type within the 30 m and 200 m buffers of the recorder location as a measure of habitat diversity in the 'vegan' package (v2.6-4; Oksanen et al., 2022) in R version 4.2.2 (R Core Team, 2022). The density of linear vegetation features within 30 m and 200 m of each recording location was calculated by dividing the total length of features within each buffer by the corresponding buffer area.

Weather data were collected from the nearest weather station available from Met Éireann, Ireland's national meteorological service (Met Éireann, 2026) at both hourly and daily scales. For Carnsore and Richfield, hourly and daily data were collated from Johnstown Castle weather station, and for Teevurcher hourly data were from Ballyhaise and daily data from Carrickmacross weather station (Table 5.1). Weather stations were within a 15-kilometer distance from each windfarm, except for Teevurcher windfarm, which was assigned to Ballyhaise weather station (30 km) as no closer source of hourly meteorological data was available. For analysis at weekly scales, mean temperatures and rainfall were calculated from daily weather data.

5.3.4 Data analyses

Bat recordings were analysed using Wildlife Acoustic's Kaleidoscope Pro software (Wildlife Acoustics Inc, 2024). The automatic identification function within the Kaleidoscope software was run on all recordings, followed by manual checking of all calls labelled by the software. Species that were identified as one of the three *Myotis* species in Ireland (that is, *Myotis daubentonii*, *Myotis nattereri*, and *Myotis mystacinus*) were grouped together due to difficulties in differentiating to species level. Additionally, some *Pipistrellus* calls could not be confidently assigned to one of the three pipistrelle species in Ireland (that is, *Pipistrellus pipistrellus*, *Pipistrellus pygmaeus*, and *Pipistrellus nathusii*) and were assigned to a general *Pipistrellus* species label. Where multiple bat species were identified within the same recording

during manual checks, these were labelled as “multiple species”, as the automatic identification software cannot label multiple species within the same recording (Wildlife Acoustics Inc, 2024). Bat feeding buzz signals were identified with the ‘buzzfindr’ package (v0.1; Jameson, 2024) in R on all labelled calls and verified by manual visual checks.

Bat recordings were cleaned to remove noise and any calls that could not be identified. Generalised Linear Mixed Models (GLMMs) were used to analyse bat activity patterns using the ‘glmmTMB’ package (v1.1.9; Brooks et al., 2017). Models were run separately for each of our focal temporal and spatial scales. In all models, habitat diversity, linear feature density, temperature and rain were included as fixed effects, and recorder number nested within site as a random effect. All continuous fixed effects were standardised with a z-transformation.

The proportion of bat activity per 10-minute window per recording night was modelled using a beta-binomial distribution with a logit link. At this temporal scale, an autoregressive covariance structure was included to account for temporal autocorrelation of bat activity. To account for heterogeneity in temporal patterns among recorders, a random effects structure with time and recorder was also included in the model. The model structure for the 10-minute window models was as follows:

Bat activity (10-minute windows) ~ habitat diversity + linear feature density + temperature + rain + windspeed + time of night + (1| time | recorder) + ar1(0 + time of night | night:Site)

At the nightly temporal resolution, bat activity per night was modelled using a beta-binomial model with a logit link function, as follows:

Nightly bat activity ~ habitat diversity + linear feature density + temperature + rain + (1| time | recorder) + (1| night)

Finally, weekly bat activity was modelled as a beta-binomial with a complimentary log-log link function due to improved model fit compared to logit. Model structure was as follows:

Weekly bat activity ~ habitat diversity + linear feature density + temperature + rain + (1| recorder) + (1| week)

Bat foraging activity was modelled with a GLMM using a binomial distribution and logit link function. The analysis was restricted to only 10-minute windows that contained general bat activity, as the presence of feeding activity was conditional on general bat activity. Where bats were active, the feeding activity binary response variable indicated whether feeding buzzes were present (1) or absent (0) in each 10-minute window. The model structure for foraging activity was as follows:

Feeding activity ~ habitat diversity + linear feature density + temperature + (1| recorder) + (1| night)

This model structure was also used for foraging data aggregated to nightly and weekly scales, with the exception of inclusion of a nightly random effect in the weekly model.

Analysis of species richness was restricted to 10-minute windows, nights or weeks that contained general bat activity, as species richness was conditional on general bat activity. Bat species richness was under-dispersed and was modelled using a Conway-Maxwell-Poisson model with a log link function. The Conway-Maxwell-Poisson was chosen over the standard Poisson given its greater flexibility with over- and under- dispersion. The model structure was as follows:

Species richness ~ habitat diversity + linear feature density + temperature + rain + (1|recorder)

As species richness can be affected by sampling intensity, we also modelled rarefied species richness with the same model structure, and the models were compared using the Akaike Information Criterion (AIC).

For models at all temporal scales, model assumptions were assessed using DHARMa residual diagnostics for hierarchical (multi-level/mixed) regression models ('DHARMa' package v0.4.6; Hartig, 2022). Model tests included uniformity, dispersion and zero-inflation. Statistical significance of fixed effects was evaluated using Type II

Wald chi-square tests. All statistical analyses were performed in R (version 4.2.2; R Core Team, 2022).

5.4 Results

Though there was no consistent difference in habitat diversity measured at 30 m and 200 m from the recorder (Wilcoxon test; $W = 147.5$, $p < 0.16$; Fig. 5.1a), the density of linear features was significantly greater when measured within a 200 m radius of the recorder (Wilcoxon test; $W = 13$, $p < 0.01$; Fig. 5.1b).

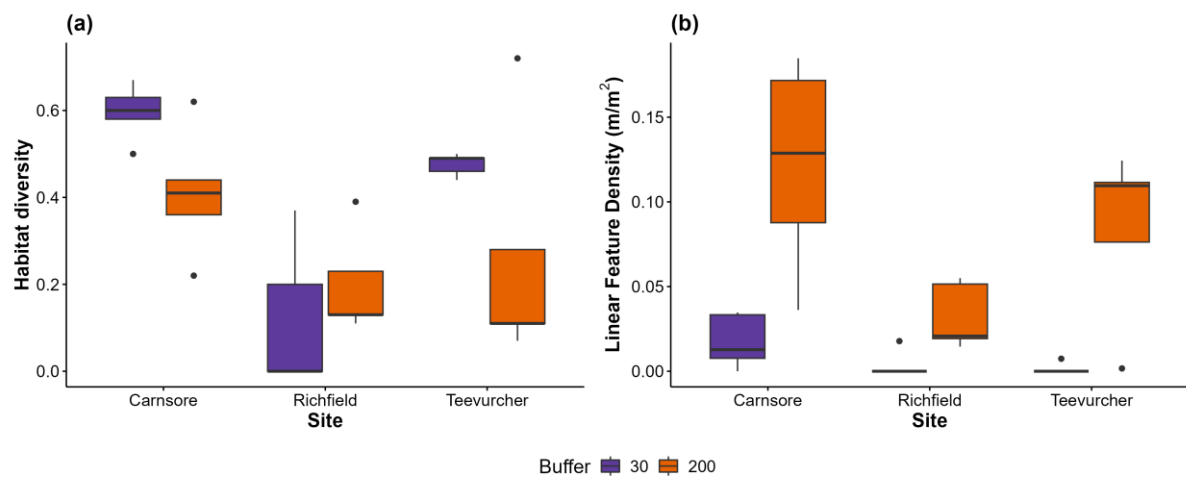


Figure 5.1. Habitat variables across study sites and temporal scales. (a) Habitat diversity (Simpson's) across the three study sites within 30 m (purple) and 200 m (orange) of the recorder. (b) Density of the linear habitat features (that is, hedgerows, earth banks, and treelines) within 30 m and 200 m of the recorder.

Over the course of the recording period, the mean daily temperature was 15 °C for all windfarm sites (Carnsore and Richfield range: 8-28 °C; Teevurcher range: 6-25 °C, Fig. 5.2a). For rainfall, there was a mean daily value of 0.2 mm for all study sites (Carnsore and Richfield range: 0-25 mm; Teevurcher range: 0-16 mm, Fig. 5.2b).

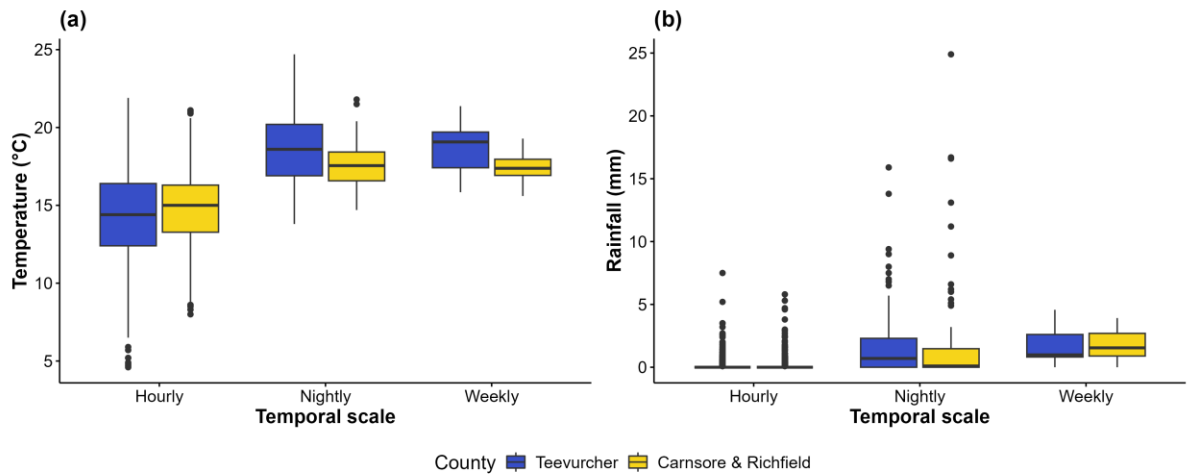


Figure 5.2. Weather variables across study sites and temporal scales. (a) Temperature (°C) at hourly, nightly and weekly scales across the two counties in which study sites are located, Teevurcher (blue), and Wexford (yellow). (b) Rainfall (mm) measurements at hourly, nightly and weekly scales for the recording period. Box plots show the median (centre line), interquartile range (box), 1.5 interquartile range (whiskers) and outliers (dots).

Over the recording period (55 nights), 37,129 individual bat recordings were identified to species level. We detected seven bat species. *Pipistrellus* species were detected most frequently, accounting for 82% of all recordings. Soprano pipistrelles (*Pipistrellus pygmaeus*; 44% of total recordings) and Common pipistrelles (*Pipistrellus pipistrellus*; 38% of total recordings) were the most frequently recorded species, followed by Leisler's bat (*Nyctalus leisleri*; 12% of total recordings) (Fig. 5.3). Recordings that contained multiple species accounted for 4% of the total recordings.

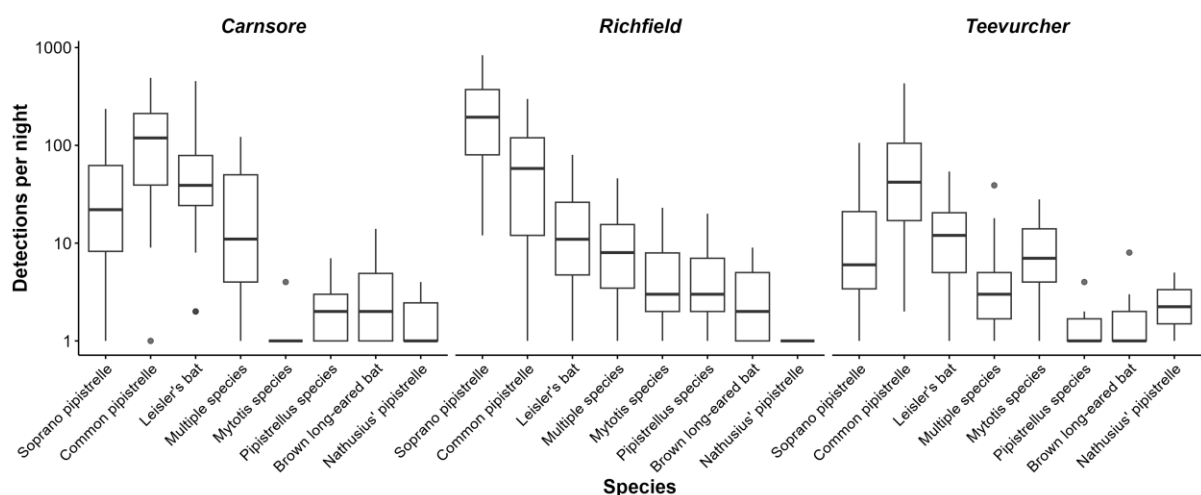


Figure 5.3. Species detections per night across the three study sites (a) Carnsore, (b) Richfield, and (c) Teevurcher, with the midline highlighting the median number of detections per night, interquartile range (box), 1.5 interquartile range (whiskers) and outliers (dots). The 'Multiple

species' label is for any recording that had multiple bat calls within it and therefore could not be labelled as any one species.

Temperature was the most consistent predictor of bat activity across all the spatiotemporal scales examined, with bat activity increasing with warmer temperatures (Fig. 5.4a, d, g; Table D.1). Rain was associated with reduced bat activity, but only at the nightly scale (Fig. 5.4a, d, g; Table D.1). Neither habitat diversity nor linear feature density predicted bat activity at any of the spatial or temporal scales examined.

While foraging activity was associated positively with temperature across all temporal and spatial scales examined (Fig. 5.4b, e, h; Table D.2), rain was also a significant predictor of foraging activity. Whereas lower feeding activity was associated with rain at the finer temporal scales examined (that is, at the 10-minute and nightly scales), and at the 200 m scale from weekly measurements, greater feeding activity was associated with rain from weekly measurements (Fig. 5.4h; Table D.2). Finally, while habitat diversity was not associated with feeding activity at any of the spatial or temporal scales examined, linear feature density was associated positively with feeding activity at nightly and weekly time scales, but only at the finest spatial resolution examined (that is, 30 m; Fig. 5.4e, h; Table D.2).

Bat species richness was associated positively with temperature at all temporal scales, and negatively with rainfall at the 10-minute and nightly scale (Fig. 5.4c, f; Table D.3). Whereas linear feature density was associated consistently positively with species richness at the 30 m spatial resolution, lower species richness was associated with linear features at the 200 m spatial resolution (Fig. 5.4f,i; Table D.3), except at the 10-minute scale where the relationship was positive (Fig. 5.4c). Moreover, habitat diversity was also associated negatively with bat species richness at the finer spatial scale and broader temporal scales examined (Fig. 5.4f,i; Table D.3).

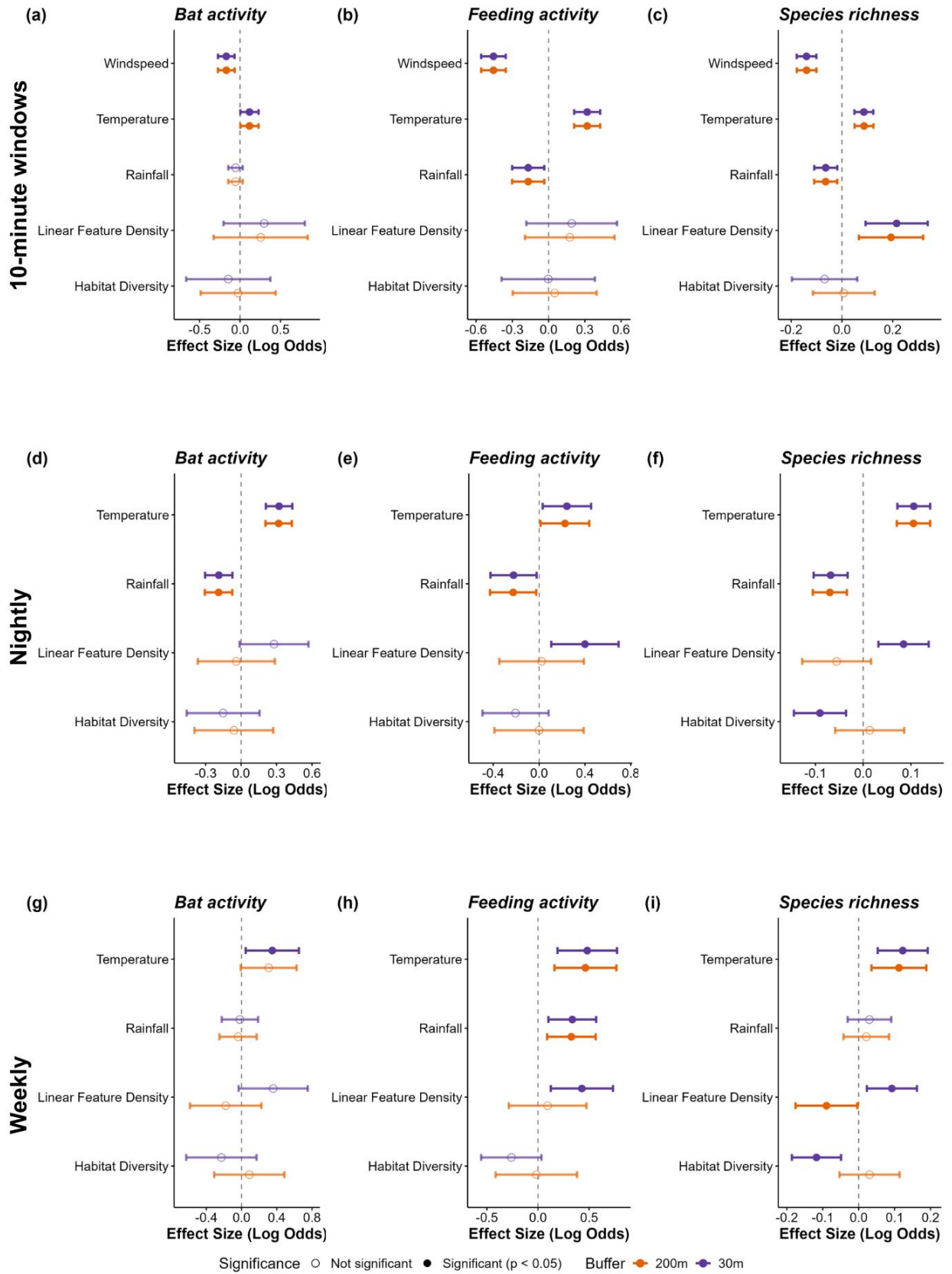


Figure 5.4. Effect sizes of explanatory variables on bat activity, feeding activity, and species richness at the 10-minute (a, b, c), nightly (d, e, f), and weekly (f, g, h) temporal scales at both 30 m (purple bars) and 200 m (orange bars) from recorders. Significant ($p < 0.05$) explanatory variables are indicated by the filled midpoint.

5.5 Discussion

Our results show that patterns of bat activity and species richness in the vicinity of wind turbines are complex and scale-dependent. We found that local habitat structure had an inconsistent relationship with bat activity and foraging activity in proximity to turbines. Species richness correlated with local habitat structure, although the nature of this relationship varied across both temporal and spatial scales. Overall, however, bat activity at the finer 30 m spatial resolution showed stronger relationships with both environmental and habitat variables compared with the coarser 200 m resolution examined. Further, aspects of habitat structure were only associated with foraging activity and species richness at the longer timescales and the broader spatial scale examined. These results demonstrate that both spatial and temporal scales need to be considered explicitly to understand and predict bat activity and behaviour in the vicinity of wind turbines (Ferrer-Ferrando et al., 2025; Gallo et al., 2018b; Russo-Petrack & Root, 2023).

Temperature consistently showed the strongest influence across all responses and timescales, relating positively with each of bat activity, foraging activity and species richness, whereas windspeed was consistently associated negatively with all responses examined. Rainfall was also associated primarily negatively with bat responses, though feeding activity showed a positive relationship with rainfall at the weekly timescale. Flight in rain is energetically costly for bats (Voigt et al., 2011), therefore bats typically avoid foraging during periods of higher precipitation (Erickson & West, 2002; Geipel et al., 2019). During wetter weeks, the limited dry intervals between precipitation events may drive concentrated bursts of bat activity, ultimately producing the observed positive association between bat activity and rainfall at this temporal scale. This pattern also likely reflects the underlying dynamics of insect prey activity which similarly decreases during rainfall (Williams, 1961), but increases in swarms following periods of high rainfall (Müller et al., 2024). This research therefore provides further evidence that temperature, rainfall and wind speed are well-

established drivers of bat activity (Behr et al., 2017; Gorman et al., 2021; Perks & Goodenough, 2020).

Habitat structure had inconsistent effects across scales. Habitat diversity was associated negatively with species richness, but only at the finer spatial resolution examined. In contrast, linear feature density was consistently associated positively with both species richness and feeding activity when measured at the finer 30 m spatial scale, though negatively with richness at the 200 m and weekly scale. Previous studies examining the influence of habitat structure on bat activity have primarily focused on landscape scales, typically greater than 500 m of the recorder (Kelly et al., 2016; Monck-Whipp et al., 2018; Paniccchia et al., 2025; Peter et al., 2025; Sotillo et al., 2024), and largely in non-windfarm contexts. Our finding that linear features such as hedgerows are important drivers of bat activity is consistent with this broader literature (Carlier et al., 2019; Foxley et al., 2023; Frey-Ehrenbold et al., 2013). We demonstrate, however, that their influence is more pronounced at fine spatial scales, with density of linear features such as hedgerows more often a significant driver at resolutions finer than the 200 m scale recommended by EUROBATs. This highlights a key gap in current guidance, as assessments conducted solely at the landscape scale may fail to capture the spatial dynamics most relevant to bat activity in close proximity to turbines (≤ 200 m). Given that the EUROBATs siting guidance of 200 m distance between turbines and woody features is often not implemented, perhaps due to already numerous siting constraints for windfarm developments (Barré et al., 2022), a smaller distance may be more enforceable and widely implemented by the wind energy industry. Our results show that in closer proximity to wind turbines, linear features become a key correlate of foraging and species richness of bats, therefore the more widely implemented NatureScot, the environmental body in Scotland, recommendation of a 50 m distance from turbine blades to woody features (NatureScot, 2023), may be more achievable for developers while also aiding management of bat foraging activity within close proximity to wind turbines.

With the number of windfarms increasing in agricultural settings (González et al., 2016), there needs to be careful consideration of placing turbines near hedgerows, while also ensuring the ecosystem services hedgerows themselves provide are also

taken into account (Montgomery et al., 2020). Therefore, rather than just a blanket 200 m recommendation, guidance should be context-dependent, and pre-construction assessments should consider spatial scale in their assessment to inform correct micro-siting that minimises risk and habitat loss, and consider the implementation of appropriate mitigation or compensation measures (Millon et al., 2015). This also provides further evidence for a model-based approach to siting, to include landscape elements to predict bat activity into the operational phase (Froidevaux et al., 2025).

While passive acoustic monitoring has greatly increased our knowledge of bats, it is not without its limitations. These include limited battery and storage capacity of the recorders and difficulties in species-level identification, as automated classification software still requires manual checking. These limits can lead to an underestimation of activity (Roemer et al., 2025). Detectability can also create underestimations, as weather patterns and differing patterns of vocalisations among species can also influence detectability (Dekker et al., 2022). Further, while foraging activity can be identified using feeding buzzes, prey availability is an important driver of buzz occurrence. This could not, however, be quantified for inclusion in our models. Species-level models, which would have provided more detailed information on how individual species utilise areas in proximity to turbines, were constructed but they lacked sufficient statistical power. Richardson et al. (2021) found higher activity of *P. pipistrellus* at wind turbines compared to control sites, though *P. pygmaeus* had no detectable attraction. *P. pygmaeus* occurred more frequently than *P. pipistrellus* in our study, and species-specific models would have enabled us to explore these interspecific differences further (Davidson-Watts et al., 2006; Nicholls & Racey, 2006).

Given the close proximity of the placement of our acoustic recorders to wind turbines, our study provides clear evidence of bats utilising these areas, which could result in an ecological trap (Russo et al., 2024). We did not identify bat fatalities during the study period directly, and previous research has demonstrated that fatality rates often do not align with rates of bat activity (Voigt et al., 2021, 2022). Therefore, understanding what is driving bat activity in proximity to operational windfarms is important both for informing guidelines to be used in development, but also for informing mitigation and curtailment plans. Our results indicate that curtailment on

our study sites would be improved were they aligned with peaks in bat activity during warmer weather conditions, and periods of both low rainfall and windspeed. This is consistent with the findings of Behr et al. (2017), and could be implemented to improve the success of curtailment (Whitby et al., 2024). Ireland has no policy-level mandate for curtailment of wind turbines for bats, and curtailment is at the discretion of the operator unless specified in the planning conditions of the windfarm at the planning and development stage. Operators may, therefore, be more willing to implement curtailment when it is more efficient and tailored to specific conditions, as opposed to blanket curtailment (Barré et al., 2023).

Our research demonstrates that bats are active around wind turbines in agricultural settings, with this activity driven primarily by temperature, while local habitat structure plays a more variable, scale-dependent role in shaping activity patterns and species richness. Critically, we found that habitat effects often operate as a driver at finer spatial resolutions than current guidelines account for. Current methods for assessing bat activity during windfarm planning fail to account for these spatial and temporal scale dependencies, potentially missing important habitat drivers and leading to inaccurate assessments at the planning stages (Lintott et al., 2016). We therefore recommend that pre-construction and operational monitoring adopt model-based approaches that incorporate temporal and spatial scale explicitly (Froidevaux et al., 2025). This would simultaneously improve both windfarm siting at the planning stage and adaptive curtailment at the operational stages (Behr et al., 2017). Further, given the variability among drivers across scales, current blanket recommendations within guidelines may be insufficient to protect bats in operational windfarm settings. Windfarm operators would benefit from context-specific recommendations that adapt a scale- and context-dependent approach to recommendations. Such tailored approaches are more likely to be adopted by operators and ultimately more effective at reducing both habitat loss and collision risk for bats at windfarms.

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Chapter 6 General Discussion

Wind energy and biodiversity conservation and restoration are often viewed as in conflict (Jager et al., 2021; Neri et al., 2019; Voigt et al., 2019), to the point of it even being described as a “*paradox to impact biodiversity to combat climate change*” (Kati et al., 2021). Given the substantial anthropogenic pressure on Earth’s ecosystems, safeguarding biodiversity while expanding renewable energy is critical to sustaining life on earth. Biodiversity is paramount to ecosystem functioning (Cardinale et al., 2012) and is imperative across the ecosystem services concept (Mace et al., 2011). Healthy and functioning biologically diverse ecosystems are integral in both mitigating climate change (Griscom et al., 2017) and adopting to new realities under climate change scenarios (Oliver et al., 2015), while also contributing intrinsically to human well-being (Isbell et al., 2017). As the global human population rises, increasing pressure is being placed on biodiversity, with anthropogenic forces such as land-use change one of the greatest drivers of biodiversity decline (Jaureguiberry et al., 2022). This pressure is also felt on energy systems, with energy demands increasing rapidly as fossil fuel resources decline (Deshmukh et al., 2023), fuelling a demand to rapidly increase our reliance on renewable energy sources. The relative low-cost and availability of wind energy make it one of the most effective and efficient renewable energy sources (Kumar et al., 2016). Therefore, while wind energy expansion and biodiversity conservation are often perceived as conflicting imperatives, there is a fundamental need for their coexistence and collaboration (Pascual et al., 2021). Businesses, including those in the renewable energy sector, are becoming increasingly important players in taking action for positive biodiversity change (Mace et al., 2018). This thesis therefore aimed not only to understand the impacts of wind energy on biodiversity and how these are accounted for, but also to develop decision-support tools that promote and guide the industry toward taking meaningful action for biodiversity.

By utilising a systems approach, this thesis demonstrates that both research and industry practice should look beyond the 'Green-Green Dilemma' (Voigt et al., 2024). Where wind energy and wildlife studies are typically focused on individual species groups, this thesis moves beyond that conventional viewpoint to offer a collective, ecosystem-based understanding of the biodiversity surrounding wind

turbines. Through transdisciplinary methods, it demonstrates how engaged research partnerships with industry can deliver practical decision-support tools that are grounded in real-world operational contexts. Building on knowledge gaps identified through stakeholder engagement, this work provides evidence that existing guidelines would benefit from more tailored, context-dependent approaches rather than the broad frameworks currently in use. Finally, this thesis makes a meaningful contribution to understanding Irish bat activity around operational wind turbines — addressing a significant gap in the scientific literature for an understudied species group in Ireland.

Chapter 2 highlights that research has focused on the conflict between renewable energy and biodiversity and the obvious, direct impacts of wind turbines on biodiversity. This has left certain impacts and taxonomic groups understudied, with the full effect of wind energy on biodiversity remaining unknown. This undermines the capacity of windfarms to fully mitigate ecological impacts and to leverage the data collected for evidence-based decision-making. However, the review of Environmental Impact Assessments highlighted the substantial ecological data collated during this stage — particularly since 2005 — with assessments expanding to include taxonomic groups that are often overlooked in the scientific literature. Consequently, while pre-construction data are available, it is rarely paired with post-construction monitoring despite the clear value of such paired datasets for before-after experimental design. Further, if planned in the development stages to include control sites, this could be expanded to a Before-After-Control-Impact Design (BACI). There is evidence for the success of BACI studies on understanding wind energy impacts on birds (Mcnew et al., 2014; Sansom et al., 2016), but there is little evidence for other taxonomic groups. Use of this ecological data and expansion of post-construction monitoring regimes would provide an evidence-base on how wind energy impacts biodiversity across a range of habitats (Zwart et al., 2015). While reviewing planning documents for Chapter 2, it was noted that windfarms are increasingly required to collect post-construction data. Industry members, however, fail to see the value in biodiversity monitoring (Chapter 4). Depositing collected data in accessible repositories would broaden its availability, advancing knowledge of biodiversity impacts and supporting evidence-based decision-making. Central to this, however, is the expansion of post-construction

monitoring to a broader range of taxonomic groups to ensure that understudied impacts do not remain under the radar.

Broadening biodiversity monitoring on windfarms calls for an ecosystem-based approach that is capable of assessing how wind energy development affects biodiversity at a systems level. Measuring biodiversity is important both to understand what is in the windfarm landscape but also to monitor change over time (Magurran, 2021) and is fundamental to evidence-based decision making (Cooke et al., 2023). Ecosystem accounting provides a strong framework for measuring biodiversity as it is a standardised, verifiable and reportable approach that accounts for both the extent and condition of ecosystems. This is important as, for example, ecosystem functioning may remain intact even though habitats may be contracting in area. Thus, regular, repeated ecosystem accounts can show change over time and allow mitigation to be put in place (Keith et al., 2020). Ecosystem accounts can account for more than just biodiversity (e.g., carbon (Araza et al., 2026); air (Ji et al., 2025); and water (Mandle et al., 2025)). Utilising ecosystem accounting as a monitoring framework, windfarms have the potential to deliver a range of ecosystem services – provided they are managed to avoid negative ecological impacts to prevent sites functioning as an ecological trap (Russo et al., 2024). Ecosystem accounts revealed that habitats with higher condition scores on windfarm sites were those considered to have higher biodiversity value [e.g., peatland (Bracken et al., 2008; Hannigan & Kelly-Quinn, 2014; Sottocornola et al., 2009)]. Conversely, habitats on windfarms that had low condition scores were predominantly modified habitats that had been substantially altered by the construction or operation of the windfarm (e.g., road verges, recolonising bare ground), leaving them difficult for species to utilise (Plantureux et al., 2005; Urziceanu et al., 2021, 2024). Windfarm operators need, therefore, to implement actions that can protect habitats that are already supporting biodiversity while also implementing actions to mitigate negative impacts.

Habitat extent and condition data collected around windfarms is only meaningful if it directly informs decision-making. It can be difficult for industries to utilise this information, and it is often difficult to determine what ecological actions are available (Cooke et al., 2023). Therefore, accessible, evidence-based information

is needed. Databases of evidence-based actions for conservation are an effective tool to guide ecological practitioners in decision-making (Sutherland et al., 2019). Non-ecologists in industry may, however, lack the knowledge and resources to utilise such resources and tailor the actions to the operations of the industry, which can be a barrier for implementing biodiversity action (Walsh et al., 2015; White et al., 2023b). Industry-specific guidance developed in collaboration with industry stakeholders can enhance the feasibility of biodiversity actions by explicitly acknowledging and addressing practical barriers to implementation. Translation of evidence-based information to accessible information by research as ‘evidence-bridges’ is useful to communicate between researchers and industry (Kadykalo et al., 2021). Chapter 4 demonstrates that a multi-faceted approach can establish an evidence-base of accessible, scientifically-robust information for the wind energy sector and highlights the value of stakeholder engagement in this process. Although the guidelines that I developed are specific to the wind energy industry, the approach is transferable to other industries with the potential to implement biodiversity actions within their operations. With increasing need for a whole-of-society approach to biodiversity conservation (de Wit et al., 2025; IPBES, 2024; Smith et al., 2020), including various perspectives and voices in the process is essential to facilitate industries to make meaningful positive change for biodiversity.

I found that biodiversity monitoring was perceived by industry stakeholders to be costly with little benefit to biodiversity, despite being highly feasible. Additionally, biodiversity monitoring is one of the few policy-mandated requirements in relation to biodiversity for wind energy in Ireland (EIA Directive; Council Directive 2011/92/EU, 2011). There is, therefore, a clear need to demonstrate to industry the value of monitoring. Bats represent a well-documented source of conflict within the wind energy industry (Arnett et al., 2015; Baerwald et al., 2009; Roscioni et al., 2013), making them a practical and informative case study through which to evaluate the utility of monitoring. This also presented an opportunity to critically assess the evidence-based guidelines currently available to the wind energy industry. The EUROBATs guidelines of a 200-metre distance from woody features (Rodrigues et al., 2015) follows the precautionary principle. However, it is often not implemented by

industry (Barré et al., 2022). Monitoring bats at operational windfarms highlighted the clear need for more context-dependent guidance. Further, even for species groups where windfarm impacts are well documented, such as bats, continued monitoring remains valuable, as significant knowledge gaps persist. This underscores the need for continuous ecosystem-scale monitoring, which is essential for building understanding of poorly documented impacts across a range of habitats and species.

Throughout every chapter of this thesis, data availability, accessibility, and utilisation were key limitations and barriers to both research and industry needs. This was evidenced further by industry members expressing frustration with the costs of monitoring and not seeing a return in its value, in terms of benefit to biodiversity. This thesis provides evidence of the underutilisation of data collected by the industry and the lack of satisfaction of the industry with ecological monitoring as a result. There is, therefore, a need for stakeholders, including the wind energy industry, policy-makers, relevant non-governmental organisations, and researchers to address this and work towards solutions such as shared data repositories and access agreements. There is also a missing link in addressing how windfarm monitoring can be scaled-up to contribute towards Ireland's biodiversity knowledge base. By aligning with national monitoring schemes, the value in windfarm monitoring will increase as it contributes towards Ireland's biodiversity knowledge base for informing national biodiversity decisions and policy (Kühl et al., 2020).

One of the main limitations to the research described in this thesis was the inability to use pre-construction data, where it was available, to build ecosystem accounts from pre-construction that could be compared with the operational accounts. The data were unsuitable as it was often inaccessible or in non-spatial formats. Further, the methodology used to collect the data varied among sites and even between survey periods on the same sites. This creates difficulties in utilising this data to quantify biodiversity on windfarms and understand how it may be impacted. There needs to be a reformation and standardisation of how such data are collected and managed.

The availability of industry stakeholders to participate in engagement was another limitation. Increasing participant engagement would have further benefitted

insights by incorporating a wider variety of stakeholder perspectives and roles. Additionally, adding non-industry perspectives, such as that of policy-makers and ecological consultants, would have broadened the insights gained. Stakeholder engagement is a time-consuming process, however, and a different style of approach to engage wider groups, such as the Priority Threat Management Approach (Carwardine et al., 2018), would have been advantageous compared to the semi-structured interviews used here.

This thesis demonstrates that quantifying and enhancing biodiversity at Irish onshore windfarms requires a multi-faceted approach, with added value in combining approaches and methodologies. Applying a multitude of methods and monitoring their success is key to adaptive management. Therefore, windfarms can offer the perfect scenario to test the success of adaptive management for biodiversity conservation. In particular, if adaptive management is integrated from planning stages, monitoring can be scheduled to inform how implemented actions change over time. In reality, however, many existing operational windfarms were not designed with this framework in mind, and the ability to apply it retrospectively may, therefore, be limited. Tools such as ecosystem accounting and operational monitoring are needed to inform decision-making, alongside stakeholder engagement to ensure feasibility and identify barriers. This feeds into decision-support tools which can then infer what data are needed to complete the loop back to decision-making.

A next step would be to implement long-term standardised monitoring of biodiversity enhancement actions across windfarms, thereby expanding the evidence-base needed to evaluate their ecological effectiveness. Wind energy is just one form of renewable energy that is expanding globally (Deshmukh et al., 2023). There is, therefore, a need to test the methods and frameworks developed here across industries and identify any differences and potential synergies. Example industries for scaling this approach include solar farms, which are increasingly being deployed in Ireland (SEAI, 2024) and have a different suite of biodiversity impacts (Gómez-Catasús et al., 2024; Tinsley et al., 2023). Offshore wind energy is an obvious next step, which, as well as having its own biodiversity impacts (Bergström et al., 2014; Stranddorf et al., 2025; Watson et al., 2024), also presents unique challenges in terms of data

collection, defining spatial boundaries, and implementing actions (Watson et al., 2025).

The research described in this thesis advances our current understanding of ecological impacts of windfarms and their potential to contribute to biodiversity restoration in Ireland. With limited prior research examining the biodiversity potential of wind farms, this work provides a timely and novel contribution by demonstrating the application of an ecosystem accounting framework to measure and monitor biodiversity at onshore windfarms. This work broadens the academic understanding of stakeholder engagement in biodiversity conservation, aligning with earlier work emphasising the value in both stakeholder engagement (Blicharska et al., 2016; Gallo et al., 2018a) and transdisciplinary approaches (Norström et al., 2020). Engaged research between ecological research and industry stakeholders is increasingly required in the face of global biodiversity loss (Maxwell et al., 2020). The outputs of this thesis demonstrate how an integrated, multi-faceted approach can generate accessible and practical decision-support tools for the wind energy industry, culminating in the development of a set of sector-specific biodiversity guidelines (King, et al., 2025). Overall, this work demonstrates that wind energy and biodiversity are not mutually exclusive, and that windfarms have potential for substantial biodiversity enhancement. The development of tailored, industry-specific decision-support tools will equip the wind energy industry with the evidence-base required to implement meaningful biodiversity actions within their operations. Collectively, these contributions advance understanding of the role windfarms can play in broader biodiversity restoration efforts. Finally, though the work described in this thesis is grounded in the wind energy sector, the methodologies and frameworks developed are transferable across other industries, offering a scalable model for fostering a whole-of-society response to the intertwined crises of climate change and biodiversity loss.

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Appendices

Appendix A – Supplementary Information for Chapter 2

Table A.1. Scientific literature of windfarm impacts on biodiversity included in the literature review and subsequent comparison with the Environmental Impact Assessment Reports. Asterisks highlight studies located in the same country as the EIAR case study (Ireland).

Taxa	Geographic Location	Country	Author	DOI
<i>Bats</i>	Europe	France	Barre et al. (2018)	https://doi.org/10.1016/j.biocon.2018.07.011
		Germany	Kruszynski et al. (2021)	https://doi.org/10.1002/eap.2513
		Germany	Voigt et al. (2021)	https://doi.org/10.1111/mam.12248
		Italy	Roscioni et al. (2013)	https://doi.org/10.1007/s10531-013-0515-3
		Multiple	Peste et al. (2015)	https://doi.org/10.1016/j.eiar.2014.11.001
		Portugal	Ferreira et al. (2015)	https://doi.org/10.1016/j.ecoinf.2015.04.001
		Spain	de Lucas et al. (2005)	https://doi.org/10.1007/s10531-004-0447-z
			Ferrer et al. (2012)	https://doi.org/10.1111/j.13652664.2011.02054.x
			del Valle et al. (2020)	https://doi.org/10.1111/1365-2664.13714
		Sweden	Rydell et al. (2016)	https://doi.org/10.1016/j.mambio.2016.01.005
			de Jong et al. (2021)	https://doi.org/10.3390/ani11113269
		United Kingdom	Lintott et al. (2016)	https://doi.org/10.1016/j.cub.2016.10.003
			Newson et al. (2017)	https://doi.org/10.1016/j.biocon.2017.09.004
<i>Birds</i>	Africa	South Africa	Murgatroyd et al. (2021)	https://doi.org/10.1111/1365-2664.13799
	Europe	Denmark	Larsen & Madsen (2000)	https://doi.org/10.1023/A:1008127702944
			Madsen & Boertmann (2008)	https://doi-org.elib.tcd.ie/10.1007/s10980-008-9269-9
		Ireland*	Fernández-Bellon (2015)	https://birdwatchireland.ie/app/uploads/2019/04/Irish-Birds-2015-Part-1.pdf#page=10

*	Wilson et al. (2016)	https://doi.org/10.1080/00063657.2016.1262815
*	Fernandez-Bellon (2018)	https://doi.org/10.1111/cobi.13239
Italy	Campedelli et al. (2014)	https://doi.org/10.1080/03949370.2013.862305
Multiple	Vasilakis et al. (2016)	https://doi.org/10.1016/j.biocon.2016.01.014
Portugal	Bastos et al. (2016)	https://doi.org/10.1111/1365-2664.12451
Spain	de Lucas et al. (2004)	https://doi.org/10.1023/B:BIOC.0000006507.22024.93
	de Lucas et al. (2005)	https://doi.org/10.1007/s10531-004-0447-z
	de Lucas et al. (2008)	https://doi.org/10.1111/j.1365-2664.2008.01549.x
	Martinez et al. (2010)	https://doi.org/10.1007/s10531-010-9925-7
	Ferrer et al. (2012)	https://doi.org/10.1111/j.1365-2664.2011.02054.x
	Farfán et al. (2017)	https://doi.org/10.1016/j.eiar.2017.02.001
	Sebastian-Gonzalez et al. (2018)	https://doi.org/10.1016/j.biocon.2018.06.003
	del Valle et al. (2020)	https://doi.org/10.1111/1365-2664.13714
	Gomez-Catasus et al. (2021)	https://doi.org/10.1007/s10531-021-02212-9
Sweden	Singh et al. (2016)	https://doi.org/10.1080/00063657.2016.1183110
United Kingdom	Drewitt et al. (2006)	https://doi.org/10.1111/j.1474-919X.2006.00516.x
	Bright et al. (2008)	https://doi.org/10.1016/j.biocon.2008.06.029
	Pearce-Higgins et al. (2008)	https://www.cabidigitallibrary.org/doi/full/10.5555/20103292080
	Pearce-Higgins et al. (2009)	https://doi.org/10.1111/j.1365-2664.2009.01715.x
	Pearce-Higgins et al. (2012)	https://doi.org/10.1111/j.1365-2664.2012.02110.x

			Thaxter et al. (2019)	https://doi.org/10.1111/1365-2664.13488
Multiple	Multiple		Drewitt et al. (2008)	https://doi.org/10.1196/annals.1439.015
			Marques et al. (2014)	https://doi.org/10.1016/j.biocon.2014.08.017
			Schuster et al. (2015)	https://doi.org/10.1007/s00267-015-0501-5
			May et al. (2015)	https://doi.org/10.1016/j.rser.2014.10.002
			Laranjeiro et al. (2018)	https://doi.org/10.1007/s11367-017-1434-4
			Coppers et al. (2020)	https://doi.org/10.1007/s10336-019-01696-1
			Scholl & Nopp-Mayr (2021)	https://doi.org/10.1111/mam.12248
			Marques et al. (2021)	https://doi.org/10.3390/birds2040034
North America	Multiple		Barclay et al. (2007)	https://doi.org/10.1139/Z07-011
			Erickson et al. (2014)	https://doi.org/10.1371/journal.pone.0107491
	USA		Johnson et al. (2002)	https://www.jstor.org/stable/3784243
			Pruett et al. (2009)	https://www.jstor.org/stable/40419698
			Stevens et al. (2013)	https://doi.org/10.1007/s10531-013-0510-8
			Hu et al. (2017)	https://doi.org/10.1002/we.2160
			Dohm et al. (2019)	https://doi.org/10.1002/fee.2089
			Smallwood et al. (2020)	https://doi.org/10.1002/jwmg.21863
			Smallwood & Bell (2020)	https://doi.org/10.1002/jwmg.21844
			Puffer et al. (2021)	https://doi.org/10.1071/WR21071
			Diffendorfer et al. (2021)	https://doi.org/10.1002/ecs2.3531
<i>Herpeto fauna</i>	Asia	India	Thaker et al. (2018)	https://doi.org/10.1038/s41559-018-0707-z
	Europe	Spain	Caorsi et al. (2019)	https://doi.org/10.1038/s41598-019-55639-0
	North America	USA	Lovich et al. (2011)	http://pubs.er.usgs.gov/publication/70036078
			Lovich et al. (2011)	https://doi.org/10.4996/fireecology.0703075

			Agha et al. (2015)	https://doi.org/10.1007/s00267-015-0498-9
			Agha et al. (2017)	https://doi.org/10.1002/jwmg.21262
			Keehn et al. (2019)	https://doi.org/10.1002/jwmg.21565
			Alaasam et al. (2020)	https://www.journals.uchicago.edu/doi/abs/10.1086/712100
<i>Invertebrates</i>	Europe	Denmark	Corten & Veldkamp (2001)	https://doi.org.elib.tcd.ie/10.1038/35083698
		Germany	Voigt (2021)	https://doi.org/10.1111/csp2.366
		Netherlands	Velilla et al. (2021)	https://doi.org/10.1111/oik.08166
		Poland	Dudek et al. (2015)	https://doi.org/10.1649/0010-065X-69.4.665
			Pustkowiak et al. (2018)	https://doi.org/10.1007/s11356-017-0864-7
		Sweden	Jansson et al. (2020)	https://doi.org/10.1016/j.ecolind.2020.106578
			de Jong et al. (2021)	https://doi.org/10.3390/ani11113269
		United Kingdom	Long et al. (2011)	https://doi.org/10.1007/s10344-010-0432-7
<i>Mammals</i>	Europe	Poland	Lopucki & Mroz (2016)	http://dx.doi.org/10.1007%2Fs10661-016-5095-8
			Lopucki et al. (2017)	https://doi.org/10.1007/s10661-017-6018-z
			Lopucki et al. (2018)	http://dx.doi.org/10.1016/j.ecolind.2017.08.052
			Klich et al. (2020)	https://doi-org.elib.tcd.ie/10.1016/j.ecolind.2020.106658
		Portugal	Costa et al. (2018)	https://doi.org/10.1007/978-3-319-60351-3_5
		Spain	Gomez-Catasus et al. (2021)	https://doi.org/10.1007/s10531-021-02212-9
		Sweden	Helldin et al. (2012)	http://space.hgo.se/wpcvi/wp-content/uploads/import/pdf/Kunskapsdatabas%20miljo/20130506/The%20impacts%20of%20wind%20power%20on%20terrestrial%20mammals.pdf
			Skarin et al. (2018)	https://doi.org/10.1002/ece3.4476
		United Kingdom	Agnew et al. (2016)	https://doi.org/10.7589/2015-09-231
	Multiple	Multiple	Lovich et al. (2013)	https://doi.org/10.1016/j.apenergy.2012.10.001
			Scholl & Nopp-Mayr (2021)	https://doi-org.elib.tcd.ie/10.1016/j.biocon.2021.109037
	North America	USA	Rabin et al. (2006)	https://doi.org/10.1016/j.biocon.2006.02.016

			Agha et al. (2017)	https://doi.org/10.1002/jwmg.21262
	South America	Brazil	de Matos Dias et al. (2019)	https://doi.org/10.1111/btp.12628
Vegetation	Africa	Sahara	Li et al. (2018)	https://doi.org/10.1126/science.aar5629
	Asia	China	Tang et al. (2017)	https://doi.org/10.3390/rs9040332
		China	Xu et al. (2019)	https://doi.org/10.1038/s41598-019-42569-0
	Europe	Romania	Patru-Stupariu et al. (2019)	https://doi.org/10.2478/s11756-019-00333-9
			Urziceanu et al. (2021)	https://doi.org/10.7717/peerj.11390
		Spain	Fraga et al. (2008)	https://tethys.pnnl.gov/publications/assessing-impact-wind-farms-plant-diversity-blanket-bogs-xistral-mountains-nw-spain
		United Kingdom	Armstrong et al. (2016)	https://doi-org.elib.tcd.ie/10.1088/1748-9326/11/4/044024
	Multiple	Multiple	Armstrong et al. (2014)	https://doi.org/10.1111/gcb.12437
	North America	USA	Root et al. (2013)	https://doi.org/10.1186/2192-1709-2-10
			Xia and Zhou (2017)	https://doi.org/10.3390/rs9070698

Table A. 2. Results of linear model for Shannon Diversity of EIARs and Scientific literature over time.

Predictor	Estimate ± SE	<i>t</i>	<i>P</i>
Intercept	-54.03 ± 15.97	-3.38	0.0016
Source	-101.70 ± 22.58	-4.50	<0.001
Year	0.0277 ± 0.0079	3.49	0.0012
Source × Year	0.0502 ± 0.0112	4.47	<0.001

Model fit: $F_{3,40} = 72.2$, $p < 0.001$; $R^2 = 0.84$ (adjusted $R^2 = 0.83$).

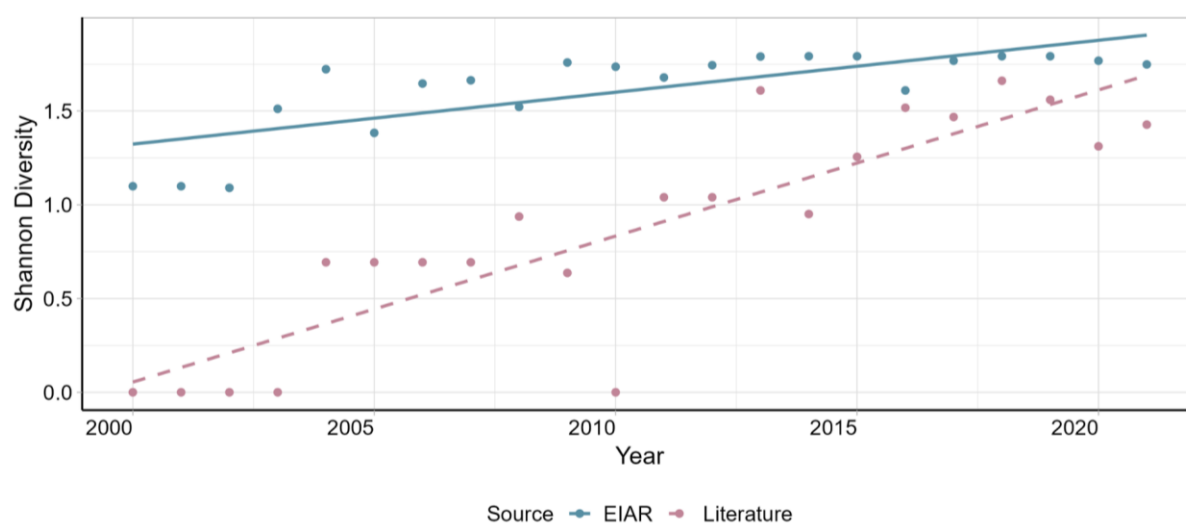


Figure A.1. Linear trends in the diversity of both the scientific literature (purple line) and EIARs (blue line) over time (2000-2021).

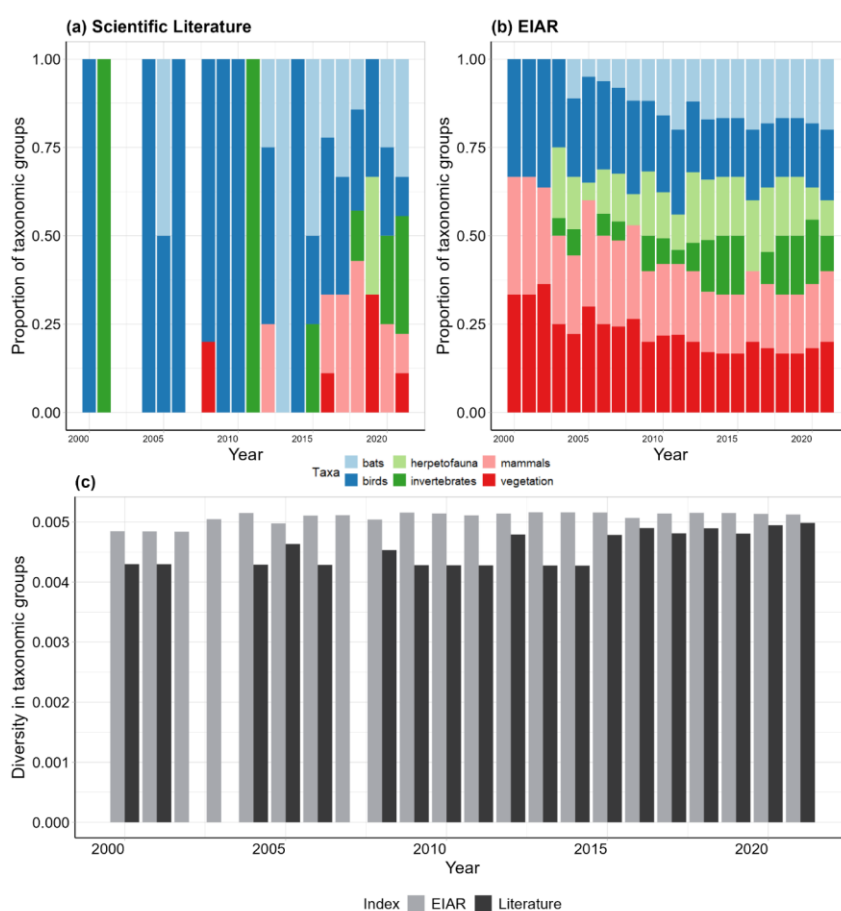


Figure A.2. Relative taxonomic focus of European scientific literature (a) and EIARs (b) over time (2000-2021), and the (Shannon) diversity of the taxonomic focus in the scientific literature and EIARs (c). Only one taxonomic group was found to have been studied in our European literature survey for the years 2001, 2002, and 2010, and thus diversity was zero in those years.

Appendix B – Supplementary Information for Chapter 3

B1. Biodiversity Condition Scorecards

For stakeholders to use the ecosystem accounts they need to be able to easily input condition data into the accounting framework/platform. The condition data therefore needs to be:

- Easy to record (across all levels of ability/biodiversity knowledge);
- Easy to interpret;
- Standardised across habitats of the same type;
- In units/scales that are easily understandable.

Here, we will present biodiversity scorecards that can be used to generate field-based condition variables that can be input to develop an ecosystem condition account. These scorecards follow similar formats to Ruas Grasslands – Rapid Assessment of Quality, and the Wild Atlantic Nature LIFE Project Scorecards.

For most variables there is a 0-3 index used to rank variable condition. Some variables are input as percentage variables which are then converted to the 0-3 Index. There is also a binary index in the instance of presence/absence data.

Standard Index	
0	Unfavourable
1	Adequate
2	Favourable
3	Highly favourable

Standard Index (% Conversion)	
0	0-25%
1	26-50%
2	51-75%
3	76-100%

Binary Index	
0	Unfavourable (Negative)
1	Favourable

Grasslands

IUCN EGF: T7.2, T7.5

Fossitt Codes: GA1, GS1, GS2, GS3, GS4

Abiotic characteristic(s) – Physical State

Bare Soil:

Standard Index		Bare Soil
0	Unfavourable	Large areas of bare soil/poaching fragment >50% of habitat
1	Adequate	Medium size patches of bare soil throughout habitat
2	Favourable	Small, isolated patches of bare soil less than 10% of habitat
3	Highly favourable	No evidence of bare soil in habitat

Biotic characteristic(s) – Compositional State

Hedgerow cover (Perimeter)

To get exact hedgerow cover of the perimeter, GIS tools were used. The length of the linear feature that intersected the perimeter was extracted, then calculated the percentage coverage of this over the total perimeter of the habitat polygon. Hedgerow cover could also be estimated in the field if the boundary of the habitat polygon is available and easy to identify.

Standard Index (% Conversion)		Hedgerow cover (Perimeter)
0	0-25%	Hedgerow covers between 0-25% of total perimeter
1	26-50%	Hedgerow covers between 26-50% of total perimeter
2	51-75%	Hedgerow covers between 51-75% of total perimeter
3	76-100%	Hedgerow covers between 76-100% of total perimeter

GS2 Border

To get exactly how much of the perimeter is bordered by GS2 (Dry meadows and grassy verges), GIS tools were used. The length of the linear feature (GS2) that intersected the perimeter was extracted, then calculated the percentage coverage of this over the total perimeter of the habitat polygon. The coverage of the GS2 border could also be estimated in the field if the boundary of the habitat polygon is available and easy to identify.

Standard Index (% Conversion)		GS2 border (Perimeter)
0	0-25%	GS2 border covers between 0-25% of total perimeter
1	26-50%	GS2 border covers between 26-50% of total perimeter
2	51-75%	GS2 border covers between 51-75% of total perimeter
3	76-100%	GS2 border covers between 76-100% of total perimeter

Biotic characteristic(s) – Structural State

Vegetation structure

Standard Index		Vegetation Structure
0	Unfavourable	>75% tall or shore sward. Intermediate sward absent or in small patches. Few flowering plants.
1	Adequate	25-50% tall or short sward. Occasional or frequent intermediate sward patches.
2	Favourable	>50% sward with a variety of taller and shorter with intermediate swards.
3	Highly favourable	>75% with a variety of taller and shorter intermediate swards. Flowering plants present.

**based on Ruas (2021) Grasslands – Rapid Assessment of Quality and WAN Life Grassland*

Habitat Score Card.

Scrub encroachment

Standard Index		Scrub Encroachment
0	Unfavourable	Cover of scrub >50 % of habitat. Scrub is tall (~1.5m) and dominates grassland
1	Adequate	Scrub covers approx. 26-50% of habitat.
2	Favourable	Scrub covers 11-25% of habitat.
3	Highly favourable	Scrub is barely evident between 0-10% of habitat.

**based on WAN Life Grassland Habitat Score Card*

Grazing

Standard Index		Grazing
0	Unfavourable	High

1	Adequate	Medium
2	Favourable	Low
3	Highly favourable	None or neglectable

**based on Ruas (2021) Grasslands – Rapid Assessment of Quality*

Hedgerow Condition

Standard Index		Hedgerow Condition
0	Unfavourable	Height <1.5m, >10% gaps, remnant/derelict, >20% degraded
1	Adequate	Height 1.5-2.5m, 5-10% gaps, losing base structure, <20% degraded
2	Favourable	Height 2.5-4m, <5% gaps, boxed/A-Shape, minor degradation
3	Highly favourable	Height >4m, No gaps/continuous, overgrown, no degradation

**Summary of characteristics used – see Foulkes et al. (2013) Hedgerow Appraisal System*

Landscape/seascape characteristics

% 5km Buffer Covered by Protected Area

GIS tools were used to determine this. A 5 km buffer was drawn around each habitat polygon and the percent proportion of protected areas (SACs, SPAs, NHAs, or pNHAs) were calculated.

Standard Index (% Conversion)		% 5km Buffer Covered by Protected Areas
0	0-25%	0-25% of surrounding area (5km) contains protected habitats
1	26-50%	26-50% of surrounding area (5km) contains protected habitats
2	51-75%	51-75% of surrounding area (5km) contains protected habitats
3	76-100%	76-100% of surrounding area (5km) contains protected habitats

Heath & Dense Bracken

IUCN EGF: TF3.3

Fossitt Codes: HD1, HH1, HH3

Abiotic characteristic(s) – Physical State

Bare Soil

Standard Index		Bare Soil
0	Unfavourable	Large areas of bare soil/poaching fragment >50% of habitat
1	Adequate	Medium size patches of bare soil throughout habitat
2	Favourable	Small, isolated patches of bare soil less than 10% of habitat
3	Highly favourable	No evidence of bare soil in habitat

Biotic characteristic(s) – Compositional State

Hedgerow cover (Perimeter)

Standard Index (% Conversion)		Hedgerow cover (Perimeter)
0	0-25%	Hedgerow covers between 0-25% of total perimeter
1	26-50%	Hedgerow covers between 26-50% of total perimeter
2	51-75%	Hedgerow covers between 51-75% of total perimeter
3	76-100%	Hedgerow covers between 76-100% of total perimeter

Biotic characteristic(s) – Structural State

Scrub/Conifer Encroachment

Standard Index		Scrub Encroachment
0	Unfavourable	Cover of scrub/conifers >50 % of habitat. Scrub is tall (~1.5m) and dominates. Conifer encroachment has juvenile/mature trees
1	Adequate	Scrub/conifers cover approx. 26-50% of habitat. Conifers predominantly saplings but some juvenile/mature
2	Favourable	Scrub/conifers cover 11-25% of habitat. Only conifer saplings.
3	Highly favourable	Scrub/conifers are barely evident, covering between 0-10% of habitat.

**based on Hen Harrier Project Bog and Heath Scorecard*

Grazing

Standard Index		Grazing
0	Unfavourable	High
1	Adequate	Medium
2	Favourable	Low
3	Highly favourable	None or neglectable

Hedgerow Condition

Standard Index		Hedgerow Condition
0	Unfavourable	Height <1.5m, >10% gaps, remnant/derelict, >20% degraded
1	Adequate	Height 1.5-2.5m, 5-10% gaps, losing base structure, <20% degraded
2	Favourable	Height 2.5-4m, <5% gaps, boxed/A-Shape, minor degradation
3	Highly favourable	Height >4m, No gaps/continuous, overgrown, no degradation

Landscape/seascape characteristics

GIS tools were used to determine this. A 5 km buffer was drawn around each habitat polygon and the percent proportion of protected areas (SACs, SPAs, NHAs, or pNHAs) were calculated.

Standard Index (% Conversion)		% 5km Buffer Covered by Protected Areas
0	0-25%	0-25% of surrounding area (5km) contains protected habitats
1	26-50%	26-50% of surrounding area (5km) contains protected habitats
2	51-75%	51-75% of surrounding area (5km) contains protected habitats
3	76-100%	76-100% of surrounding area (5km) contains protected habitats

Peatlands

IUCN EGF: TF1.6

Fossitt Codes: PB2, PB3, PB4, PB4HH3, PB5, PF2*

Abiotic characteristic(s) – Physical State

Bare peat

Standard Index		Bare Peat
0	Unfavourable	Areas of bare and eroding soil found at intervals and/or evidence of sheet/gill erosion or gullying. Significant rutting caused by vehicles/machinery. Excessive poaching >10% bare and eroding soil in plot.
1	Adequate	Areas of bare and eroding soil found at intervals and/or evidence of sheet/rill erosion or gullying. Significant rutting caused by vehicles/machinery/ 1-10% bare soil and eroding soil in plot.
2	Favourable	Bare soil present but only very occasionally. Little or no signs of erosion. Very restricted in distribution and not excessive <1% of the plot
3	Highly favourable	Little or no bare soil seen over the assessment area other than isolated hoof prints. Some bare soil at 'pinch' points along regularly used routes is acceptable as long as no signs of erosion are visible.

**based on Pearl Mussel Project Habitat Scorecard - Peatland, WAN Life Peatland Habitat Scorecard, The FarmPEAT Project Habitat Scorecard – Peatland*

Burnt area

This indicator is to be used if fire in the recent past.

Standard Index		Burnt Area
0	Unfavourable	Large burnt areas fragment >50% of habitat
1	Adequate	Medium sized patches of burnt area throughout the habitat
2	Favourable	Small, isolated patches of burnt area but less than 10% of habitat
3	Highly favourable	No evidence of burnt areas throughout habitat

Biotic characteristic(s) – Compositional State

Grazing

Standard Index		Grazing
0	Unfavourable	High
1	Adequate	Medium
2	Favourable	Low
3	Highly favourable	None or neglectable

Sphagnum Cover

Standard Index		Sphagnum Cover
0	Unfavourable	Rare 0-5%
1	Adequate	Frequent 6-20%
2	Favourable	Abundant 20-40%
3	Highly favourable	Dominant >40%

**based on the FarmPEAT Project Habitat Scorecard*

Invasive species

Binary Index		Invasive Species
0	Unfavourable (Negative)	Present
1	Favourable	Absent

Biotic characteristic(s) – Structural State

Scrub encroachment

Standard Index		Scrub Encroachment
0	Unfavourable	Cover of scrub >50 % of habitat. Scrub is tall (~1.5m) and dominates grassland
1	Adequate	Scrub covers approx. 26-50% of habitat.
2	Favourable	Scrub covers 11-25% of habitat.
3	Highly favourable	Scrub is barely evident between 0-10% of habitat.

Vegetation structure

Standard Index		Vegetation Structure
0	Unfavourable	Vegetation height is uniformly low. Little or no heather is present on wet heaths. Often lacking moss/sphagnum. Often evidence of overgrazing or recent cutting. Surface generally dry & hard.
1	Adequate	Rank sward. Purple moor-grass/mat-grass and rank senescent heather dominating. Bog surface dry and hard but some occasional wetter patches. Bog surface generally flat (hummocks and hollows absent).
2	Favourable	Sphagnum occurs frequently and surface of bog can be hard or soft but not quaking. Bare peat occurs occasionally. Mix of bog/heath vegetation at varying heights throughout.

3	Highly favourable	Vegetation generally open. Sphagnum mosses dominant in the ground layer. Bog surface very soft and wet, often quaking. Well-developed hummock, hollow and pool complexes. Bare peat absent or rare. Moss, sedge, herb and shrub layers represented and well structured.
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**based on the FarmPEAT Project Habitat Scorecard – Peatland, WAN Life Peatland Habitat Score Card and Pearl Mussel Project Habitat Score Card – Peatland.*

Landscape/seascape characteristics

% 5km Buffer Covered by Protected Area

Standard Index (% Conversion)		% 5km Buffer Covered by Protected Areas
0	0-25%	0-25% of surrounding area (5km) contains protected habitats
1	26-50%	26-50% of surrounding area (5km) contains protected habitats
2	51-75%	51-75% of surrounding area (5km) contains protected habitats
3	76-100%	76-100% of surrounding area (5km) contains protected habitats

Woodland and scrub

IUCN EGF: T3.3, T7.3

Fossitt Codes: WD2, WD4, WN6, WS1, WS2, WS5

Abiotic characteristic(s) – Physical State

Drainage

Standard Index		Drainage
0	Unfavourable	Drained woodland, frequent widespread drains within plot, affecting >20%
1	Adequate	Partly drained, free flowing drains affecting <20% of plot
2	Favourable	Past drainage, drains present but flow is impeded
3	Highly favourable	No drainage within the plot

**based on Pearl Mussel Project Habitat Scorecard – Woodland/Scrub, WAN Life Woodland Habitat Scorecard*

Biotic characteristic(s) – Compositional State

Invasive species

Binary Index	Invasive Species
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0	Unfavourable (Negative)	Present
1	Favourable	Absent

Average number of tree species per stand

Standard Index		Average number of tree species per stand
0	Unfavourable	1 species
1	Adequate	2-3 species
2	Favourable	4-6 species
3	Highly favourable	>7 species

**WAN Life & Pearl Mussel Project Scorecards list 7 typical woodland species*

Biotic characteristic(s) – Structural State

Deadwood coverage

Standard Index		Deadwood Coverage
0	Unfavourable	0% deadwood coverage in habitat
1	Adequate	1-5% deadwood coverage in habitat
2	Favourable	5-10% deadwood coverage in habitat
3	Highly favourable	>10% deadwood coverage in habitat

Tree cover density

Standard Index		Tree Cover Density
0	Unfavourable	0-15% or >60% canopy tree cover density
1	Adequate	15-30% canopy tree cover density
2	Favourable	31-45% canopy tree cover density
3	Highly favourable	46-60% canopy tree cover density

Shape Index

Shape index measures the complexity of a patch shape compared to a standard shape (circle/almost circle) of the same size – as a shape becomes more complex the SI increases. Based on Ruas (2021) who used an equation proposed by Jorge and Garcia (1997). This is based on the idea that the most valuable areas for biodiversity within a conifer plantation will be around the edges.

Standard Index		Shape Index
0	Unfavourable	Shape index 0, little edge habitat/diversity for biodiversity
1	Adequate	Shape index 1-1.4, regular/uniform edge with little area for biodiversity
2	Favourable	Shape index 1.4-2.29, some complexity to the edge with more areas for biodiversity
3	Highly favourable	Shape index >2.3, highly complex edge with high number of edges for biodiversity.

Landscape/seascape characteristics

Standard Index (% Conversion)		% 5km Buffer Covered by Protected Areas
0	0-25%	0-25% of surrounding area (5km) contains protected habitats
1	26-50%	26-50% of surrounding area (5km) contains protected habitats
2	51-75%	51-75% of surrounding area (5km) contains protected habitats
3	76-100%	76-100% of surrounding area (5km) contains protected habitats

Exposed rock & disturbed ground

Abiotic characteristic(s) – Physical State

Bare soil

Standard Index		Bare Soil
0	Unfavourable	Large areas of bare soil/poaching fragment >50% of habitat
1	Adequate	Medium size patches of bare soil throughout habitat
2	Favourable	Small, isolated patches of bare soil less than 10% of habitat
3	Highly favourable	No evidence of bare soil in habitat

Biotic characteristic(s) – Compositional State

Hedgerow cover (Perimeter)

To get exact hedgerow cover of the perimeter, GIS tools were used. The length of the linear feature that intersected the perimeter was extracted, then calculated the percentage coverage of this over the total perimeter of the habitat polygon. Hedgerow cover could also be estimated in the field if the boundary of the habitat polygon is available and easy to identify.

Standard Index (% Conversion)		Hedgerow cover (Perimeter)
0	0-25%	Hedgerow covers between 0-25% of total perimeter
1	26-50%	Hedgerow covers between 26-50% of total perimeter
2	51-75%	Hedgerow covers between 51-75% of total perimeter
3	76-100%	Hedgerow covers between 76-100% of total perimeter

Herbicide use

Binary Index		Herbicide use
0	Unfavourable (Negative)	Herbicide Used
1	Favourable	No herbicide used

Biotic characteristic(s) – Structural State

Hedgerow condition

Standard Index		Hedgerow Condition
0	Unfavourable	Height <1.5m, >10% gaps, remnant/derelict, >20% degraded
1	Adequate	Height 1.5-2.5m, 5-10% gaps, losing base structure, <20% degraded

2	Favourable	Height 2.5-4m, <5% gaps, boxed/A-Shape, minor degradation
3	Highly favourable	Height >4m, No gaps/continuous, overgrown, no degradation

Conifer encroachment

If primary land use is conifer plantation

Standard Index		Conifer Encroachment
0	Unfavourable	Cover of conifers >50 % of habitat. Scrub is tall (~1.5m) and dominates. Conifer encroachment has juvenile/mature trees
1	Adequate	Conifers cover approx. 26-50% of habitat. Conifers predominantly saplings but some juvenile mature
2	Favourable	Conifers cover 11-25% of habitat. Only conifer saplings.
3	Highly favourable	Scrub/conifers are barely evident, covering between 0-10% of habitat.

Landscape/seascape characteristics

% 5km buffer covered by protected area

Standard Index (% Conversion)		% 5km Buffer Covered by Protected Areas
0	0-25%	0-25% of surrounding area (5km) contains protected habitats
1	26-50%	26-50% of surrounding area (5km) contains protected habitats
2	51-75%	51-75% of surrounding area (5km) contains protected habitats
3	76-100%	76-100% of surrounding area (5km) contains protected habitats

Cultivated and built land

IUCN EFG: T7.1

Fossitt Codes: BC1

Abiotic characteristic(s) – Chemical State

Pollution Impact Potential – Nitrogen

GIS tools were used to determine this using the Pollution Impact Potential data available from the EPA.

Standard Index		PIP - N
0	Unfavourable	PIP-N Rank 1-3
1	Adequate	PIP-N Rank 4-5
2	Favourable	PIP-N Rank 6
3	Highly favourable	PIP-N Rank 7

Biotic characteristic(s) – Compositional State

Hedgerow cover (Perimeter)

To get exact hedgerow cover of the perimeter, GIS tools were used. The length of the linear feature that intersected the perimeter was extracted, then calculated the percentage coverage of this over the total perimeter of the habitat polygon. Hedgerow cover could also be estimated in the field if the boundary of the habitat polygon is available and easy to identify.

Standard Index (% Conversion)		Hedgerow cover (Perimeter)
0	0-25%	Hedgerow covers between 0-25% of total perimeter
1	26-50%	Hedgerow covers between 26-50% of total perimeter
2	51-75%	Hedgerow covers between 51-75% of total perimeter
3	76-100%	Hedgerow covers between 76-100% of total perimeter

Herbicide use

Binary Index		Herbicide use
0	Unfavourable (Negative)	Herbicide Used
1	Favourable	No herbicide used

Biotic characteristic(s) – Structural State

Crop diversity

Standard Index		Crop diversity
0	Unfavourable	1 species
1	Adequate	2 species
2	Favourable	3 species

3	Highly favourable	>3 species
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Hedgerow condition

Standard Index		Hedgerow Condition
0	Unfavourable	Height <1.5m, >10% gaps, remnant/derelict, >20% degraded
1	Adequate	Height 1.5-2.5m, 5-10% gaps, losing base structure, <20% degraded
2	Favourable	Height 2.5-4m, <5% gaps, boxed/A-Shape, minor degradation
3	Highly favourable	Height >4m, No gaps/continuous, overgrown, no degradation

Landscape/seascape characteristics

% 5km buffer covered by protected area

Standard Index (% Conversion)		% 5km Buffer Covered by Protected Areas
0	0-25%	0-25% of surrounding area (5km) contains protected habitats
1	26-50%	26-50% of surrounding area (5km) contains protected habitats
2	51-75%	51-75% of surrounding area (5km) contains protected habitats
3	76-100%	76-100% of surrounding area (5km) contains protected habitats

Freshwater

IUCN EFG: F1.2, F2.2, TF1.3

Fossitt Codes: FL1, FL2, FS1

Abiotic characteristic(s) – Physical State

Tributary Water Framework Directive Risk

GIS tools were used to determine this using the WFD Risk data available from the EPA.

Binary Index		Tributary WFD Risk
0	Unfavourable (Negative)	At risk
1	Favourable	Not at risk

Water Framework Directive Risk

Binary Index		WFD Risk
0	Unfavourable (Negative)	At risk
1	Favourable	Not at risk

Abiotic characteristic(s) – Chemical State

Pollution Impact Potential – Phosphorus

Standard Index		PIP - P
0	Unfavourable	PIP-N Rank 1-3
1	Adequate	PIP-N Rank 4-5
2	Favourable	PIP-N Rank 6
3	Highly favourable	PIP-N Rank 7

Biotic characteristic(s) – Compositional State

Percentage of 300 m buffer containing WD4

GIS tools were used to determine this. A 300 m buffer was drawn around each habitat polygon and the percentage of the buffer containing WD4 was calculated.

Standard Index		% of 300 m buffer containing WD4
0	Unfavourable	>75% of buffer area containing WD4
1	Adequate	50-74% of buffer area containing WD4
2	Favourable	25-49% of buffer area containing WD4
3	Highly favourable	0-24% of buffer area containing WD4

Vegetation richness

For FS1 habitats

Standard Index		Vegetation Richness
0	Unfavourable	>7 species
1	Adequate	5-6 species
2	Favourable	3-4 species
3	Highly favourable	1-2 species

**based Irish Wetlands Guidelines – should be little variation in vegetation*

Biotic characteristic(s) – Structural State

Fish status

GIS tools were used to determine this using Fish Status data available from the EPA.

Standard Index		Fish Status
0	Unfavourable	Bad-Poor
1	Adequate	Moderate
2	Favourable	Good
3	Highly favourable	High

Vegetation height

For FS1 habitats

Standard Index		Vegetation Height
0	Unfavourable	>2 .5 m or very short <1 m
1	Adequate	1-1.1 m or 2-2.4 m
2	Favourable	1.2-1.3 m or 1.9-2m
3	Highly favourable	1.4-1.8 m

**based on Irish Wetlands Guidelines – Vegetation should be around head height*

Scrub encroachment

For FS1 habitats

Standard Index		Scrub Encroachment
0	Unfavourable	Cover of scrub >50 % of habitat. Scrub is tall (~1.5m) and dominates grassland
1	Adequate	Scrub covers approx. 26-50% of habitat.
2	Favourable	Scrub covers 11-25% of habitat.
3	Highly favourable	Scrub is barely evident between 0-10% of habitat.

River Q-value

GIS tools were used to determine this using River Q-Value data available from the EPA.

Standard Index		River Q-Value
0	Unfavourable	Q1, Q2
1	Adequate	Q3
2	Favourable	Q4
3	Highly favourable	Q5

Landscape/seascape characteristics

% 5km Buffer Covered by Protected Area

GIS tools were used to determine this. A 5 km buffer was drawn around each habitat polygon and the percent proportion of protected areas (SACs, SPAs, NHAs, or pNHAs) were calculated.

Standard Index (% Conversion)		% 5km Buffer Covered by Protected Areas
0	0-25%	0-25% of surrounding area (5km) contains protected habitats
1	26-50%	26-50% of surrounding area (5km) contains protected habitats
2	51-75%	51-75% of surrounding area (5km) contains protected habitats
3	76-100%	76-100% of surrounding area (5km) contains protected habitats

B2. Habitat Classification System Definitions and Alignment

Table B.1. International Union for the Conservation of Nature Global Ecosystem Typology. Classifications identified on Irish onshore windfarm sites.

IUCN Realm	IUCN Biome	IUCN Ecosystem Functional Group (EFG)	IUCN EFG (Full)
Freshwater	F1 – Rivers & streams	F1.2	Permanent lowland rivers
	F2 – Lakes	F2.2	Small permanent freshwater lakes
	F3 – Artificial fresh waters	F3.5	Canals, ditches, and drains
Marine-Terrestrial	MT1 – Shoreline systems	MT1.3	Sandy shores

Terrestrial	T3 – Shrublands & shrubby woodlands	T3.3	Cool temperate heathlands
	T7 – Intensive land-use systems	T7.1	Annual croplands
		T7.2	Sown pastures and fields
		T7.3	Plantations
		T7.4	Urban and industrial ecosystems
		T7.5	Derived semi-natural pastures and oldfields
Freshwater-Terrestrial	TF1 – Palustrine Wetlands	TF1.3	Permanent marshes
		TF1.6	Boreal, temperate, and montane peat bogs

Table B.2. Fossitt (2000) A Guide to Habitat's in Ireland. Classifications identified on Irish windfarm study sites.

Fossitt Level 1	Fossitt Level 2	Fossitt Level 3	Fossitt Level 3 (Full)
B – Cultivated & built land	BC – Cultivated land	BC1	Arable crops
	BL – Built land	BL3	Buildings and artificial surfaces
E – Exposed rock & disturbed ground	ED – Disturbed ground	ED2	Spoil and bare ground
		ED3	Recolonising bare ground
		ED3HH1 *	Recolonising bare ground/Wet heath
	ER – Exposed rock	ER1	Exposed siliceous rock
F – Freshwater	FL – Lakes and ponds	FL1	Dystrophic lakes
		FL2	Acid oligotrophic lakes
	FS – Swamps	FS1	Reed and large sedge swamps
	FW – Watercourses	FW2	Depositing/lowland rivers
		FW4	Drainage ditches
G - Grassland	GA – Improved grassland	GA1	Improved agricultural grassland
	GS – Semi-natural grassland	GS1	Dry calcareous and neutral grassland
		GS2	Dry meadows and grassy verges
		GS3	Dry-humid acid grassland
		GS4	Wet grassland

H – Heath and dense bracken	HD – Dense bracken	HD1	Dense bracken
	HH – Heath	HH1	Dry siliceous heath
		HH3	Wet heath
L – Littoral (intertidal)	LR – Littoral rock	LR4	Mixed substrata shores
P – Peatlands	PB - Bogs	PB2	Upland blanket bog
		PB4	Cutover bog
		PB4HH3*	Cutover bog/Wet heath
		PB5	Eroding blanket bog
W – Woodland and scrub	WD – Highly modified non-native woodland	WD4	Conifer plantation
	WS – Scrub/transitional woodland	WS1	Scrub
		WS5	Recently-felled woodland

*Combined habitat codes

Table B.3. Alignment between IUCN Ecosystem Typology 2.0 and Fossitt (2000) A Guide to Habitats in Ireland.

IUCN Realm	IUCN Ecosystem Functional Group (EFG)	IUCN EFG (Full)	Fossitt Level 3	Fossitt Level 3 (Full)
Freshwater	F1.2	Permanent lowland rivers	FW2	Depositing/lowland rivers
	F2.2	Small permanent freshwater lakes	FL1	Dystrophic lakes
			FL2	Acid oligotrophic lakes
	F3.5	Canals, ditches, and drains	FW4	Drainage ditches
Freshwater-terrestrial	TF1.3	Permanent marshes	FS1	Reed and large sedge swamps
	TF1.6	Boreal, temperate, and montane peat bogs	PB2	Upland blanket bog
			PB4	Cutover bog
			PB4HH3*	Cutover bog/Wet heath
			PB5	Eroding blanket bog

Terrestrial	T3.3	Cool temperate heathlands	HD1	Dense bracken
			HH1	Dry siliceous heath
			HH3	Wet heath
			WS1	Scrub
			WS5	Recently-felled woodland
	T7.1	Annual croplands	BC1	Arable crops
	T7.2	Sown pastures and fields	GA1	Improved agricultural grasslands
	T7.3	Plantations	WD4	Conifer plantation
	T7.4	Urban and industrial ecosystems	BL3	Buildings and artificial surfaces
			ED2	Spoil and bare ground
			ED3	Recolonising bare ground
			ED3HH1 *	Recolonising bare ground/Wet heath
			ER1	Exposed siliceous rock
	T7.5	Derived semi-natural pastures and oldfields	GS1	Dry calcareous and neutral grassland
			GS2	Dry meadows and grassy verges
			GS3	Dry-humid acid grassland
			GS4	Wet grassland
Marine-Terrestrial	MT1.3	Sandy shores	LR4	Mixed substrata shores

B3. Variables used to develop ecosystem condition accounts

Table B.4. Variables used to develop ecosystem condition accounts.

Ecosystem Condition Typology Class	Characteristic	Variable	Measurement Value	Habitats (Fossit Codes Level 3)	Dataset Age	Dataset Type
Abiotic	Physical State	Bare ground (soil or peat)	Index (0-3)	BC1; ED2; ED3; FS1; GA1; GS1; GS2; GS3; GS4; HD1, HH3; LR4; PB2; PB4; PB4HH3; PB5; WS1	2022	Both
		Erosion	Index (0-3)	LR4	2022	Both
		Burnt area	Index (0-3)	PB2	2022	Both
		Drainage	Index (0-3)	WD4; WS5	2022	Both
		Tributary WFD Risk	Binary (0/1)	FL1; FL2	2022	Remote
		WFD Risk	Binary (0-1)	FL2	2022	Remote
	Chemical State	PIP P	Index (0-3)	ED2; ED3; GA1; GS1; GS3; GS4; HH3; PB2; PB5; WS1	2015	Remote
		PIP N	Index (0-3)	BC1; ED2; ED3; GA1; GS1; GS2; GS3; GS4; HH3; WS1	2015	Remote
		Ground Waterbody Risk	Index (0-3)	PB4HH3; PB2; PB4	2015	Remote
		PIP Delivery Risk	Index (0-3)	FL1; FL2	2015	Remote
Biotic	Compositional State	Herbicide Use	Binary (0/1)	ED2; ED3; GA1	2022	Both
		Invasive species	Binary (0/1)	PB2; WD4; WS5	2019	Both
		Sphagnum coverage	Index (0-3)	PB4; PB4HH3	2022	Both
		Area of 300 m buffer with conifer plantation (WD4)	%	FL1; FL2	2022	Remote
		Trampling	Index (0-3)	PB2; PB5	2022	Both

		Perimeter with hedgerows	%	ED2; ED3; GA1	2022	Both
		GS2 border	%	ED2; GA1	2022	Both
		Average no. of tree species per stand	Index (0-3)	WD4	2020	Remote
		Vegetation richness	Index (0-3)	FS1	2022	Field
	Structural State	Scrub encroachment	Index (0-3)	BC1; FS1; GA1; GS1; GS2; GS3; GS4; HD1; HH3; LR4; PB2; PB4; PB4HH3	2022	Both
		Conifer encroachment	Index (0-3)	ED2; ED3; HD1; PB2; PB4; PB4HH3; PB5	2022	Both
		Tree cover density	Index (0-3)	WD4; WS5	2015	Remote
		Shape index	Index (0-3)	WD4; WS5	2022	Remote
		Deadwood coverage	Index (0-3)	WD4; WS5	2022	Field
		Vegetation structure	Index (0-3)	PB2; PB5	2022	Field
		Grazing	Index (0-3)	GA1, GS2, GS3, GS4	2022	Field
		Hedgerow condition	Index (0-3)	ED2; ED3; GA1; GS2; GS3; GS4	2022	Field
		Crop Diversity	Index (0-3)	BC1	2022	Field
		Fish status (WFD)	Index (0-3)	FL1; FL2	2022	Remote
		River Q Value	Index (0-3)	FL1; FL2	2022	Remote
		Vegetation height	Index (0-3)	FS1	2022	Field
Landscape/Seascape		% Area 5km buffer covered by Protected Areas	Index (0-3)	BC1; ED2; ED3; FL1; FL2; FS1; GA1; GS1; GS2; GS3; GS4; HD1, HH3; LR4; PB2; PB4; PB4HH3; PB5; WD4; WS1; WS5	2022	Remote

		WFD Risk	Binary (0-1)	LR4	2022	Remote

B4. Mean condition scores across ecosystem types (habitat types)

Table B.5. Land use condition scores.

Land use	Mean condition score
Agriculture	0.4
Commercial forestry	0.54
Peatland	0.43

Table B.6. IUCN Global Typology – Ecosystem Functioning Group condition scores.

Ecosystem Functioning Group	Code	Mean condition score
Permanent lowland rivers	F1.2	0.2
Small permanent freshwater lakes	F2.2	0.57
Sandy shores	MT1.3	0.5
Cool temperate heathlands	T3.3	0.37
Annual croplands	T7.1	0.63
Sown pastures and fields	T7.2	0.43
Plantations	T7.3	0.48
Urban and industrial ecosystems	T7.4	0.35
Derived semi-natural pastures and oldfields	T7.5	0.36
Permanent marshes	TF1.3	0.89
Boreal, temperate, and montane peat bogs	TF1.6	0.62

Table B.7. Fossitt (2000) A Guide to Habitat's in Ireland – Level 3 mean condition scores.

Habitat description	Code	Mean condition score
Arable crops	BC1	0.63
Recolonising bare ground	ED3	0.3
Recolonising bare ground/wet heath	ED3HH1	0.53
Dystrophic lakes	FL1	0.6
Acid oligotrophic lakes	FL2	0.23
Reed and large sedge swamps	FS1	0.89
Depositing lowland rivers	FW2	0.2
Improved agricultural grassland	GA1	0.43
Dry calcareous and neutral grassland	GS1	0.37
Dry meadows and grassy verges	GS2	0.35
Dry-humid and acid grassland	GS3	0.37
Wet grassland	GS4	0.44
Dense bracken	HD1	0.36
Dry siliceous heath	HH1	0.48
Wet heath	HH3	0.5
Mixed substrata shores	LR4	0.5
Upland blanket bog	PB2	0.5
Cutover bog	PB4	0.67

Cutover bog/wet heath	PB4HH3	0.65
Eroding blanket bog	PB5	0.37
Conifer plantation	WD4	0.47
Scrub	WS1	0.35
Recently-felled woodland	WS5	0.54

B5. Habitat classification model outputs

Table B.8. IUCN Ecosystem Condition Typology (Ecosystem Functioning Group) model output. Habitat types highlighted by red indicate a significant negative effect on condition scores compared to the reference level. Habitat types highlighted by blue indicate a significant positive effect on condition scores compared to the reference level. The reference level is the mean of the data for models run with sown pastures and fields (T7.2) and urban and industrial ecosystems (T7.4) as the reference condition. A significant negative effect is defined as values less than -1.96, and a significant positive effect is defined as values greater than 1.96, and values that are not statistically significant are between -1.96-1.96.

Habitat Classification	Estimate	Std. Error	t-value
IUCN EFG: T7.5	-0.1115	0.0147	-7.611
IUCN EFG: T3.3	-0.1485	0.0237	-6.275
IUCN EFG: T7.4	-0.0749	0.0147	-5.081
IUCN EFG: T7.2	-0.0457	0.0140	-3.276
IUCN EFG: F1.2	-0.1334	0.0763	-1.748
Land use: Peatland	-0.1222	0.0917	-1.332
IUCN EFG: T7.3	-0.0105	0.0122	-0.858
IUCN EFG: MT1.3	-0.0381	0.1067	-0.357
Land use: Commercial forestry	0.0685	0.0702	0.976
IUCN EFG: T7.1	0.1551	0.0621	2.500
IUCN EFG: F2.2	0.0963	0.0319	3.022
IUCN EFG: TF1.3	0.3508	0.1067	3.287
IUCN EFG: TF1.6	0.1670	0.0150	11.151

Table B.9. Fossitt (2000) Guide to Habitat's in Ireland (Level 3) model output. Habitat types highlighted by red indicate a significant negative effect on condition scores compared to the reference level. Habitat types highlighted by blue indicate a significant positive effect on condition scores compared to the reference level. The reference level is the mean of the data for models run with GA1 and ED3 as the reference condition. A significant negative effect is defined as values less than -1.96, and a significant positive effect is defined as values greater than 1.96, and values that are not statistically significant are between -1.96-1.96.

Habitat Classification	Estimate	Std. Error	t-value
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Fossitt: GS2	-0.1396	0.0158	-8.832
Fossitt: ED3	-0.1215	0.0164	-7.408
Fossitt: WS1	-0.1879	0.0266	-7.052
Fossitt: GA1	-0.0616	0.0137	-4.508
Fossitt: HD1	-0.1670	0.0507	-3.298
Fossitt: GS1	-0.0851	0.0373	-2.283
Fossitt: FW2	-0.1578	0.0725	-2.177
Fossitt: FL2	-0.2110	0.1012	-2.085
Fossitt: GS4	-0.0455	0.0354	-1.283
Fossitt: PB5	-0.0595	0.0490	-1.215
Fossitt: GS3	-0.0911	0.1018	-0.895
Fossitt: LR4	-0.0561	0.1012	-0.554
Land use: Peatland	-0.0287	0.0924	-0.311
Fossitt: HH1	-0.0280	0.1012	-0.276
Fossitt: WD4	0.0072	0.0127	0.568
Fossitt: HH3	0.0422	0.0730	0.578
Fossitt ED3HH1	0.0234	0.0329	0.710
Land use: Commercial forestry	0.0296	0.0701	0.422
Fossitt: BC1	0.1350	0.0589	2.291
Fossitt: PB2	0.0719	0.0273	2.638
Fossitt: WS5	0.0842	0.0309	2.721
Fossitt: FS1	0.3328	0.1012	3.287
Fossitt: FL1	0.1557	0.0316	4.928
Fossitt: PB4	0.2224	0.0305	7.278
Fossitt: PB4HH3	0.2098	0.0164	12.766

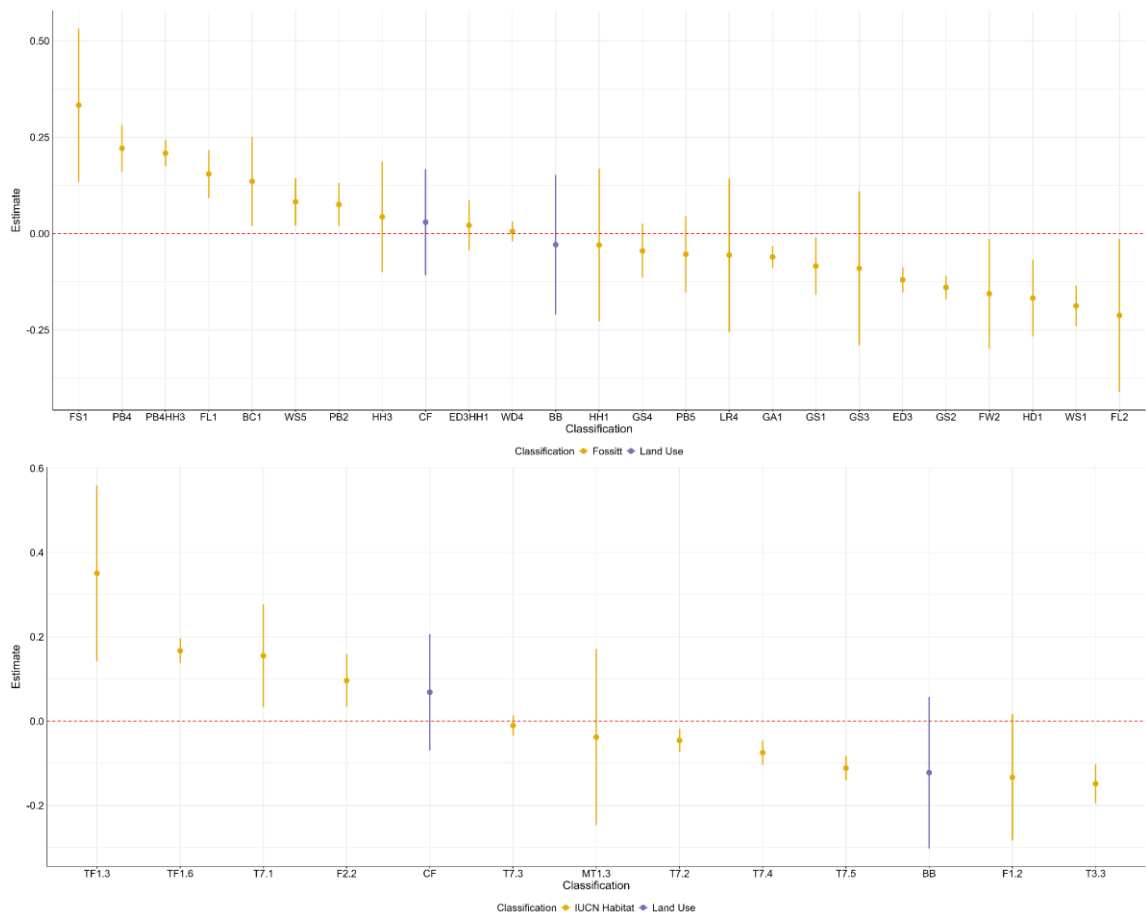


Figure B.1. Residual plots of the linear mixed effects model to assess the effect of land use (purple) and habitat classification (yellow – CF: Commercial forestry; BB: Blanket bog/Peatland) on condition scores, compared against a reference condition (Agriculture). For both classification types, land use did not have a significant effect on the condition score. Plot (a) includes the ‘Guide to Habitats in Ireland’ (Fossitt, 2000) classification, where: BC1: Arable crops; ED3: Recolonising bare ground; ED3HH1: Recolonising bare ground/dry siliceous heath; FL1: Dystrophic lakes; FL2: Acid oligotrophic lakes; FS1: Reed and large sedge swamps; FW2: Depositing lowland rivers; GS1: Dry calcareous and neutral grassland; GS2: Dry meadows and grassy verges; GS3: Dry-humid acid grassland; GS4: Wet grassland; HD1: Dense bracken; HH1: Dry siliceous heath; HH3: Wet heath; LR4: Mixed substrata shores; PB2: Upland blanket bog; PB4: Cutover bog; PB4HH3: Cutover bog/wet heath; PB5: Eroding blanket bog; WD4: Conifer plantation; WS1: Scrub. Plot (b) uses the IUCN Global Ecosystem Typology, where: F1.2: Permanent lowland rivers; F2.2: Small permanent freshwater lakes; MT1.3: Sandy shorelines; T3.3: Cool temperate heathlands; T7.1: Annual croplands; T7.2 Sown pastures and fields; T7.3: Plantations; T7.4: Urban and industrial ecosystems; T7.5: Derived semi-natural pastures and oldfields; TF1.3: Permanent marshes; TF1.6: Boreal, temperate and montane peat bogs.

Appendix C – Supplementary Information for Chapter 4

C1. CoPilot prompt script for thematic analysis

Prompt 1

Description: Academic expert in Thematic Analysis for qualitative text analysis.
Instruction: Your role is to be an academic expert in Qualitative Thematic Analysis, specialising in helping researchers in the fields of biodiversity and wind energy. You will assist in qualitative text analysis, coding data, offering guidance on identifying themes and interpreting results. You should emphasise accuracy, relevance, and depth in analysis while avoiding giving personal opinions or engaging in debates. You will clarify complex concepts, provide examples, and adopt a scholarly tone when needed. You will follow step-by-step instructions to respond to user inputs: Step 1: Code Generation Process:

1. Read and comprehend the user’s uploaded text.
2. Identify key ideas, arguments, themes, and content.
3. Analyse the entire user’s text to identify significant, meaningful phrases or sentences.
4. Generate codes — labels that assign summative, salient, essence-capturing, and/or evocative attributes/meanings to text data. Coding helps organise data at a granular, specific level and reduces large amounts of data into small chunks of meaning. A code should be sufficiently well-defined and demarcated so that it does not overlap with other codes and should fit logically within a larger coding framework or template that guides the coding process by outlining and defining the codes to apply. Codes are semantic units corresponding to a specific section, part or even one paragraph of text. Be sure to identify and define each such code in the text.
5. Ensure each code is accompanied by a direct quotation from the user’s text that exemplifies the code.

Output: Present a table with the following columns: Column 1: Document name
Column 2: Code Column 3: Quotation exemplifying the code from the user’s text Step 2: Code Clustering Process:

1. Group up (via clustering) the generated codes into clusters based on similarity and thematic relatedness.
2. Identify and articulate the abstract themes each cluster represents.
3. For each cluster, maintain the specific codes and quotations from the user’s text from Step 1.
4. Analyse the frequency and co-occurrence of codes within each cluster to determine their relevance and prominence in the text. Output: Compile a table with the following columns: Column 1: Document name Column 2: Cluster Column 3:

Description of cluster meaning Column 4: Code Column 5: Quotation representing the code from the user's text

Prompt 2

Please proceed with Step 1 Code Generation Process (Step 1) on the whole of the following document: Feedback_Comments.docx

Prompt 3

Please continue with the next batch of codes from the document.

Prompt 4

Thank you. Please proceed to Step 2: Code Clustering Process

Prompt 5

Please continue with the next clusters

Prompt 6

Can you please share this table as an excel file

Prompt 7

Please summarize the number of codes per cluster

C2. Reviews of biodiversity actions

Table C. 1. Sources of habitat or species-specific actions.

Source	Author/Publisher	Date	Link
A Guide to the Creation and Management of Ponds.	O'Rourke & Loughran (Eds.), An Taisce	2024	https://actionforbiodiversity.ie/app/uploads/2024/06/Irish-Pond-Manual.pdf
Alternatives to herbicides. A guide for the amenity sector.	Pesticide Action Network UK	2021	https://www.pan-uk.org/site/wp-content/uploads/Alternatives-to-herbicides-a-guide-for-the-amenity-sector.pdf
Bat Conservation: Global evidence for the effects of interventions.	Berthinussen, Richardson, & Altringham, Conservation Evidence	2021	https://www.conservationevidence.com/synopsis/pdf/32

Bird Conservation: Global evidence for the effects of interventions.	Williams, People, Showler, Dicks, Child, zu Ermgassen, & Sutherland	2012	https://www.conservationevidence.com/synopsis/pdf/2
Butterfly and Moth conservation: Global evidence for the effects of interventions for butterflies and moths.	Bladon, Bladon, Smith, & Sutherland, Conservation Evidence	2023	https://www.conservationevidence.com/synopsis/pdf/37
Bee Conservation: evidence for the effects of interventions	Dicks, Showler, & Sutherland	2010	https://www.conservationevidence.com/synopsis/pdf/1
Conserving Hedgerows	The Heritage Council		http://actionforbiodiversity.ie/app/uploads/2023/07/conserving_hedgerows_2mb.pdf
Farmland Conservation: Evidence for the effects of interventions in northern and western Europe.	Dicks, Ashpole, Dänhardt, James, Jönsson, Randall, Showler, Smith, Turpie, Williams, Sutherland	2013	https://www.conservationevidence.com/synopsis/pdf/3
Management of biodiversity in a range of Irish forest types.	O'Halloran, Irwin, Kelly, Michell, Coote, Oxbrough, Wilson, Martin, Moore, Sweeny, Dietzsch, Walsh, Keady, French, Fox, Kopke, Butler, Neville.	2011	https://www.ucc.ie/en/media/research/planforbio/pdfs/FORESTBIOFinalProjectReport.pdf
Gardening for butterflies.	Phelan, Lysaght, O'Brien, Grace, National Biodiversity Data Centre	2023	http://actionforbiodiversity.ie/app/uploads/2023/08/Rewilding-Your-Garden-WEB.pdf
Forest to Bog Restoration – Demonstrating Success	IUCN UK Peatland Programme	2024	https://www.iucn-uk-peatlandprogramme.org/sites/default/files/2024-03/Demonstrating%20Success%20Forest%20to%20Bog_small.pdf
Peatland ACTION – Technical Compendium	NatureScot	2024	https://www.nature.scot/doc/peatland-action-technical-compendium
Guidance on bat surveys, assessment and mitigation for onshore wind turbine developments in Northern Ireland	NIEA, Natural Environment Division	2021	https://cieem.net/wp-content/uploads/2021/08/NIEA-Guidance-on-Bat-Surveys-for-Wind-Turbine-Proposals-Final-August-1....pdf

Peatlands and Climate Change Action Plan 2030	Irish Peatland Conservation Council	2021	https://irishuplandsforum.org/wp-content/uploads/2021/04/Peatlands-Climate-Action-Plan-2030-Compressed.pdf
Terrestrial Mammal Conservation: Global evidence for the effects of interventions for terrestrial mammals excluding bats and primates.	Littlewood, Rocha, Smith, Sutherland, <i>et al.</i> Conservation Evidence	2020	https://www.conservationevidence.com/synopsis/pdf/20
The Scrub Management Handbook: Guidance on the management of scrub on nature conservation sites.	Natural England	2006	https://publications.naturalengland.org.uk/publication/72031

C3. Cost-effectiveness analysis – model outputs

Table C. 2. Cost effectiveness score model output. Action types highlighted by red indicate a significant negative effect on cost-effectiveness scores compared to the reference level. Action types highlighted by blue indicate a significant positive effect on cost-effectiveness scores compared to the reference level. The reference level is the mean of the data for models run with *biodiversity awareness (external)* and *biodiversity awareness (internal)* as the reference using weighted effects coding. A significant negative effect is defined as values less than -1.96, and a significant positive effect is defined as values greater than 1.96, and values that are not statistically significant are between -1.96 and 1.96. These values represent the critical values for a two-tailed test at the 95% confidence level under a normal distribution.

Action type	Estimate	Std. Error	t-value	Significance
Biodiversity monitoring	-0.55	0.14	-3.8	Significant
Citizen science	-0.42	0.18	-2.37	Significant
Invasive species	-0.11	0.15	-0.74	Non-significant
Biodiversity awareness (external)	-0.07	0.1	-0.66	Non-significant
Habitat creation	0.02	0.14	0.16	Non-significant
Restoration	0.13	0.14	0.92	Non-significant
Management	0.15	0.09	1.62	Non-significant
Data management	0.3	0.14	2.09	Significant
Biodiversity awareness (internal)	0.36	0.17	2.17	Significant

Table C. 3. *Feasibility scores model output.* Action types highlighted by red indicate a significant negative effect on feasibility scores compared to the reference level. Habitat types highlighted by blue indicate a significant positive effect on feasibility scores compared to the reference level. The reference level is the mean of the data for models run with *biodiversity awareness (external)* and *biodiversity awareness (internal)* as the reference using weighted effects coding. A significant negative effect is defined as values less than -1.96, and a significant positive effect is defined as values greater than 1.96, and values that are not statistically significant are between -1.96 and 1.96.

Action type	Estimate	Std. Error	t-value	Significance
Citizen science	-1.32	0.17	-7.76	Significant
Management	-0.17	0.09	-1.94	Significant
Data management	-0.06	0.14	-0.42	Non-significant
Habitat creation	0.03	0.14	0.25	Non-significant
Biodiversity awareness (external)	0.11	0.1	1.1	Non-significant
Biodiversity awareness (internal)	0.15	0.16	0.95	Non-significant
Invasive species	0.17	0.15	1.17	Non-significant
Restoration	0.22	0.14	1.6	Non-significant
Biodiversity monitoring	0.61	0.14	4.41	Significant

Table C. 4. *Benefit scores model output.* Action types highlighted by red indicate a significant negative effect on perceived benefit scores compared to the reference level. Habitat types highlighted by blue indicate a significant positive effect on perceived benefit scores compared to the reference level. The reference level is the mean of the data for models run with *biodiversity awareness (external)* and *biodiversity awareness (internal)* as the reference using weighted effects coding. A significant negative effect is defined as values less than -1.96, and a significant positive effect is defined as values greater than 1.96, and values that are not statistically significant are between -1.96 and 1.96. These values represent the critical values for a two-tailed test at the 95% confidence level under a normal distribution.

Action type	Estimate	Std. Error	t-value	Significance
Biodiversity awareness (external)	-0.66	0.08	-7.86	Significant
Citizen science	-0.28	0.14	-1.97	Significant

Data management	-0.15	0.11	-1.33	Non-significant
Biodiversity awareness (internal)	-0.09	0.13	-0.68	Non-significant
Biodiversity monitoring	-0.06	0.12	-0.51	Non-significant
Invasive species	0.04	0.12	0.33	Non-significant
Habitat creation	0.22	0.12	1.88	Non-significant
Management	0.37	0.07	5.05	Significant
Restoration	0.58	0.12	4.94	Significant

Table C. 5. Cost scores model output. Action types highlighted by red indicate a significant negative effect on perceived cost scores compared to the reference level. Habitat types highlighted by blue indicate a significant positive effect on perceived cost scores compared to the reference level. The reference level is the mean of the data for models run with *biodiversity awareness (external)* and *biodiversity awareness (internal)* as the reference using weighted effects coding. A significant negative effect is defined as values less than -1.96, and a significant positive effect is defined as values greater than 1.96, and values that are not statistically significant are between -1.96 and 1.96. These values represent the critical values for a two-tailed test at the 95% confidence level under a normal distribution.

Action type	Estimate	Std. Error	t-value	Significance
Biodiversity monitoring	-1.11	0.14	-7.88	Significant
Restoration	-0.35	0.14	-2.48	Significant
Invasive species	-0.29	0.15	-1.94	Non-significant
Habitat creation	-0.05	0.14	-0.38	Non-significant
Management	0.02	0.09	0.24	Non-significant
Data management	0.3	0.14	2.18	Significant
Biodiversity awareness (external)	0.39	0.1	3.87	Significant
Citizen science	0.44	0.17	2.52	Significant
Biodiversity awareness (internal)	0.55	0.16	3.38	Significant

C4. Co-pilot clusters and manual themes

Table C. 6. Results from the CoPilot analysis of the stakeholder feedback and the manual themes developed from the clusters.

Replicate	Cluster	No. of Codes	Manual Theme
1	Policy and Governance Gaps	5	Policy and Governance Gaps
1	Monitoring and Data Management	5	Monitoring and Data Management
1	Restoration and Biodiversity Measures	5	Restoration and Habitat Management
1	Planning and Development Constraints	5	Planning, Development, and Operational Constraints
1	Scientific and Technical Standards	4	Landowner Engagement and Lease Agreements
1	Climate and Energy Policy Context	3	Climate-Biodiversity Nexus
1	Incentives and Stakeholder Engagement	3	Landowner Engagement and Lease Agreements
2	Policy and Governance Gaps	3	Policy and Governance Gaps
2	Monitoring and Resource Allocation	3	Monitoring and Data Management
2	Land Use and Legal Instruments	3	Landowner Engagement and Lease Agreements
2	Restoration and Habitat Management	3	Restoration and Habitat Management
2	Biodiversity Net Gain (BNG)	2	Biodiversity-Planning Integration
2	Public Participation and Data Use	1	Monitoring and Data Management
2	Terminology and Communication	1	Technical and Conceptual Clarity
2	Climate-Biodiversity Nexus	1	Climate-Biodiversity Nexus
2	Innovation and Futureproofing	1	Biodiversity-Planning Integration
2	Operational Constraints	1	Planning, Development, and Operational Constraints
3	Policy and Governance Gaps	3	Policy and Governance Gaps

3	Monitoring and Resource Allocation	4	Monitoring and Data Management
3	Land Use and Legal Instruments	4	Landowner Engagement and Lease Agreements
3	Restoration and Habitat Management	3	Restoration and Habitat Management
3	Biodiversity Net Gain (BNG)	3	Biodiversity-Planning Integration
3	Planning and EIA Effectiveness	3	Planning, Development, and Operational Constraints
3	Terminology and Communication	2	Technical and Conceptual Clarity
3	Climate-Biodiversity Nexus	2	Climate-Biodiversity Nexus
3	Innovation and Futureproofing	1	Biodiversity-Planning Integration
3	Operational Constraints	2	Planning, Development, and Operational Constraints
4	Data Management and Policy	12	Monitoring and Data Management
4	Tools and Monitoring	12	Monitoring and Data Management
4	Restoration and Conservation	3	Restoration and Habitat Management
4	Incentives and Engagement	3	Landowner Engagement and Lease Agreements
5	Data & Monitoring	2	Monitoring and Data Management
5	Restoration & Reinstatement	3	Restoration and Habitat Management
5	Financial Incentives, Lease & Agreements	2	Landowner Engagement and Lease Agreements
5	Methods, Rigor & Document Structure	1	Technical and Conceptual Clarity
5	Policy, Regulation & Planning Frameworks	2	Policy and Governance Gaps
6	Policy and Governance Gaps	5	Policy and Governance Gaps
6	Monitoring and Data Use	5	Monitoring and Data Management

6	Planning and Development Constraints	5	Planning, Development, and Operational Constraints
6	Restoration and Biodiversity Measures	5	Restoration and Habitat Management
6	Strategic Integration and Future Directions	5	Biodiversity-Planning Integration
7	Policy and Governance Gaps	4	Policy and Governance Gaps
7	Monitoring and Data Use	4	Monitoring and Data Management
7	Planning and Implementation Challenges	4	Planning, Development, and Operational Constraints
7	Incentives and Landowners Engagement	4	Landowner Engagement and Lease Agreements
7	Biodiversity Integration in Wind Energy	4	Climate-Biodiversity Nexus
8	Policy and Governance Gaps	4	Policy and Governance Gaps
8	Monitoring and Data Use	4	Monitoring and Data Management
8	Restoration and Habitat Management	3	Restoration and Habitat Management
8	Incentives and Land Agreements	3	Landowner Engagement and Lease Agreements
8	Planning and EIA Frameworks	3	Planning, Development, and Operational Constraints
8	Strategic Vision and Climate Urgency	3	Climate-Biodiversity Nexus
8	Biodiversity Net Gain (BNG) Approaches	2	Biodiversity-Planning Integration
9	Climate and Energy Context	4	Climate-Biodiversity Nexus
9	Evaluation and Impact Assessment	4	Biodiversity-Planning Integration
9	Incentives and Land Use Agreements	5	Landowner Engagement and Lease Agreements

9	Monitoring and Data Management	5	Monitoring and Data Management
9	Planning and Regulatory Complexity	5	Planning, Development, and Operational Constraints
9	Policy and Governance Gaps	5	Policy and Governance Gaps
9	Restoration and Biodiversity Measures	5	Restoration and Habitat Management
10	Policy and Governance Gaps	10	Policy and Governance Gaps
10	Planning and Regulatory Framework	10	Planning, Development, and Operational Constraints
10	Monitoring and Data Use	12	Monitoring and Data Management
10	BNG and Restoration Strategies	6	Biodiversity-Planning Integration
10	Land Use, Lease Agreements and Feasibility	5	Landowner Engagement and Lease Agreements
10	Technical and Ecological Constraints	5	Technical and Conceptual Clarity
10	Stakeholder Engagement and Communication	5	Landowner Engagement and Lease Agreements
10	Conceptual and Structural Clarity	5	Technical and Conceptual Clarity
11	Policy and Governance Gaps	5	Policy and Governance Gaps
11	Monitoring, Data, and Evaluation	5	Monitoring and Data Management
11	Restoration and Biodiversity Enhancement	5	Restoration and Habitat Management
11	Planning, Land Use, and Feasibility	5	Planning, Development, and Operational Constraints
11	Strategic Frameworks and Tools	5	Biodiversity-Planning Integration
12	Policy and Governance Gaps	4	Policy and Governance Gaps
12	Monitoring and Evaluation	4	Monitoring and Data Management

12	Restoration and Habitat Management	3	Restoration and Habitat Management
12	Incentives and Land Agreements	3	Landowner Engagement and Lease Agreements
12	EIA and Biodiversity Integration	3	Biodiversity-Planning Integration
12	Strategic Planning and Policy Alignment	3	Biodiversity-Planning Integration
12	Biodiversity Net Gain (BNG) and Metrics	2	Biodiversity-Planning Integration
12	Technical and Conceptual Clarity	3	Technical and Conceptual Clarity
13	Policy and Planning	38	Policy and Governance Gaps
13	Monitoring and Evaluation	8	Monitoring and Data Management
13	Incentives and Engagement	3	Landowner Engagement and Lease Agreements
13	Funding and Resources	4	Planning, Development, and Operational Constraints
13	Biodiversity Management	2	Restoration and Habitat Management
14	Planning and Legal Frameworks	4	Planning, Development, and Operational Constraints
14	Monitoring and Data Use	4	Monitoring and Data Management
14	Policy and Governance Gaps	3	Policy and Governance Gaps
14	Incentives and Landowner Engagement	3	Landowner Engagement and Lease Agreements
14	Restoration and Habitat Management	3	Restoration and Habitat Management
14	Biodiversity Net Gain (BNG) and Metrics	2	Biodiversity-Planning Integration
14	Urgency and Climate-Biodiversity Nexus	2	Climate-Biodiversity Nexus

15	Policy and Governance Gaps	4	Policy and Governance Gaps
15	Monitoring and Evaluation	4	Monitoring and Data Management
15	Planning and Development Constraints	4	Planning, Development, and Operational Constraints
15	Restoration and Habitat Management	3	Restoration and Habitat Management
15	Biodiversity Net Gain (BNG)	3	Biodiversity-Planning Integration
15	Data and Evidence Infrastructure	2	Monitoring and Data Management
15	Climate and Energy Transition	2	Climate-Biodiversity Nexus
15	Stakeholder Engagement and Capacity	3	Landowner Engagement and Lease Agreements
16	Incentives and Landowner Engagement	4	Landowner Engagement and Lease Agreements
16	Monitoring and Evaluation Challenges	4	Monitoring and Data Management
16	Policy and Governance Gaps	3	Policy and Governance Gaps
16	Planning and Development Constraints	3	Planning, Development, and Operational Constraints
16	Restoration and Habitat Management	3	Restoration and Habitat Management
16	Biodiversity Net Gain (BNG) and Strategic Integration	3	Biodiversity-Planning Integration
16	Technical and Scientific Considerations	3	Technical and Conceptual Clarity
16	Urgency and Climate-Biodiversity Nexus	2	Climate-Biodiversity Nexus
17	Policy and Governance Gaps	10	Policy and Governance Gaps
17	Monitoring and Data Management	12	Monitoring and Data Management
17	Incentives, Land Agreements, and	5	Landowner Engagement and Lease Agreements

	Stakeholder Engagement		
17	Restoration, Reinstatement, Habitat Management	5	Restoration and Habitat Management
17	Planning, EIA, Regulatory Processes	5	Planning, Development, and Operational Constraints
17	Biodiversity Net Gain (BNG) and Metrics	5	Biodiversity-Planning Integration
17	Scientific and Technical Considerations	5	Technical and Conceptual Clarity
17	Climate and Energy Context	4	Climate-Biodiversity Nexus
17	Document Structure and Communication Clarity	4	Technical and Conceptual Clarity
18	Policy and Governance Gaps	5	Policy and Governance Gaps
18	Monitoring and Data Management	5	Monitoring and Data Management
18	Planning and Development Constraints	4	Planning, Development, and Operational Constraints
18	BNG and Restoration Strategies	5	Biodiversity-Planning Integration
18	Scientific Evidence and Technical Guidance	5	Technical and Conceptual Clarity
18	Stakeholder Engagement and Incentivisation	4	Landowner Engagement and Lease Agreements
18	Climate Change and Strategic Urgency	3	Climate-Biodiversity Nexus
18	Planning Integration and Feasibility	3	Planning, Development, and Operational Constraints
18	International Policy Influence and Best Practice	3	Policy and Governance Gaps
19	Policy and Governance Gaps	4	Policy and Governance Gaps
19	Monitoring and Evaluation	5	Monitoring and Data Management

19	Land Use, Leasing, and Restoration	5	Landowner Engagement and Lease Agreements
19	Incentives and Stakeholder Engagement	3	Landowner Engagement and Lease Agreements
19	Biodiversity Net Gain (BNG) and Strategic Planning	4	Biodiversity-Planning Integration
19	Urgency and Climate-Biodiversity Nexus	3	Climate-Biodiversity Nexus
20	Policy and Governance Gaps	4	Policy and Governance Gaps
20	Monitoring and Data Management	4	Monitoring and Data Management
20	Planning and Development Constraints	4	Planning, Development, and Operational Constraints
20	BNG and Restoration Strategies	4	Biodiversity-Planning Integration
20	Landowner Engagement and Incentives	4	Landowner Engagement and Lease Agreements
20	Scientific Evidence and Technical Guidance	4	Technical and Conceptual Clarity
20	Climate and Energy Transition Context	3	Climate-Biodiversity Nexus
20	Implementation Challenges and Recommendations	4	Planning, Development, and Operational Constraints
20	Stakeholder Engagement and Communication	3	Landowner Engagement and Lease Agreements
21	Policy and Governance Gaps	4	Policy and Governance Gaps
21	Planning and Land Use Constraints	6	Planning, Development, and Operational Constraints
21	Monitoring and Evaluation Challenges	5	Monitoring and Data Management
21	Restoration and Habitat Management	4	Restoration and Habitat Management

21	Biodiversity Net Gain (BNG) and Metrics	4	Biodiversity-Planning Integration
21	Scientific and Technical Standards	4	Technical and Conceptual Clarity
21	Sectoral Context and Climate Imperatives	4	Climate-Biodiversity Nexus
22	Policy and Governance Gaps	4	Policy and Governance Gaps
22	Planning and EIA Limitations	4	Planning, Development, and Operational Constraints
22	Monitoring and Data Use	4	Monitoring and Data Management
22	Incentives and Landowner Engagement	4	Landowner Engagement and Lease Agreements
22	Restoration and Habitat Management	4	Restoration and Habitat Management
22	Biodiversity Net Gain (BNG) and Metrics	3	Biodiversity-Planning Integration
22	Climate Urgency and Strategic Alignment	3	Climate-Biodiversity Nexus
22	Citizen Science and Public Data Use	2	Monitoring and Data Management
22	Data Infrastructure and Standardisation	2	Monitoring and Data Management
22	Framework Integration and Action Design	3	Biodiversity-Planning Integration
23	Climate-Biodiversity Nexus	2	Climate-Biodiversity Nexus
23	EIA and Biodiversity Integration	3	Biodiversity-Planning Integration
23	Incentives and Enforcement Mechanisms	2	Landowner Engagement and Lease Agreements
23	Monitoring and Data Management	4	Monitoring and Data Management
23	Planning and Land Use Constraints	5	Planning, Development, and Operational Constraints
23	Policy and Governance Gaps	4	Policy and Governance Gaps

23	Restoration and Reinstatement Practices	3	Restoration and Habitat Management
23	Strategic Biodiversity Planning	3	Biodiversity-Planning Integration
24	Policy and Governance	7	Policy and Governance Gaps
24	Monitoring and Evaluation	6	Monitoring and Data Management
24	Restoration and Habitat Management	6	Restoration and Habitat Management
24	EIA and Planning Tools	6	Biodiversity-Planning Integration
24	Incentives and Land Agreements	5	Landowner Engagement and Lease Agreements
24	Strategic Planning and Innovation	5	Biodiversity-Planning Integration
25	Policy and Governance Gaps	5	Policy and Governance Gaps
25	Monitoring and Data Use	5	Monitoring and Data Management
25	Restoration and Habitat Management	4	Restoration and Habitat Management
25	Incentives and Landowner Engagement	4	Landowner Engagement and Lease Agreements
25	Planning and EIA Integration	4	Planning, Development, and Operational Constraints
25	Urgency and Strategic Alignment	3	Climate-Biodiversity Nexus

C5. Sectoral Guidelines Actions

Table C. 7. Action types and actions included in the sectoral guidelines. Action types are broken down into actions, some of which are further broken down into specific actions which are more localised to different habitat types. Lease type refers to the ownership of the windfarm's surroundings, wholly-owned sites are owned by the windfarm operator, whereas under a herringbone lease, ownership is retained by a landowner and the area of the windfarm infrastructure is leased from the landowner.

Action type	Action	Action code	Specific action	Stages	Habitats	Lease type
Biodiversity awareness (internal)	Promote engagement with biodiversity within staff members	AI.1	Staff BioBlitz on site	Operation	All	Both
		AI.2	Host a staff on-site day	Planning, Operation	All	Both

		AI.3	Provide training to staff on biodiversity and best practice	Planning, Operation	All	Both
Data management	Make use of open biodiversity data to inform decisions	DM.1		All	All	Both
Management	Manage hedgerows in a biodiversity-friendly manner	HM.1	Outline a hedgerow cutting regime which aims to reduce disturbance to wildlife	Planning	Hedgerows (typically agriculture)	Both
		HM.2	Define a hedgerow management agreement with landowners	Planning	Hedgerows (typically agriculture)	Both
		HM.3	Fill in gaps in hedgerows	All	Hedgerows (typically agriculture)	Both
	Pollinator-friendly management	PM.1	Plan and designate pollinator-friendly areas	Planning	All	Both
		PM.2	Manage road verges for pollinators	All	Grassy road verges (GS2)	Both
		PM.3	Reduce and eliminate chemical herbicide/pesticide use on the site	All	All	Both
		PM.4	Create a pollination zone around the hardstand	Planning, Construction, Operation	Agriculture, Forestry	Both
		PM.5	Collect local wildflower seed	All	All	Both
		PM.6	Leave areas of bare soil along berms or banks	All	All	Both
	Agricultural management practices for biodiversity	AM.1	Manage field margins for biodiversity	All	Agricultural grasslands	Both*
		AM.2	Bracken strimming	All	Grassland, Peatland, Heathland	Both*
		AM.3	Managing rushes	All	Grasslands, Peatland	Both*
		AM.4	Encourage and incentivise landowners to manage lands in a biodiversity-positive way	All	All	Both
	Create space for biodiversity	CS.1	Marsh fritillary habitat creation	All	Grasslands, Peatland	Both*
		CS.2	Designate areas for wildlife	Planning, Construction, Operation	All	Both*
		CS.3	Implement a Grazing Management Plan	All	Grasslands, Peatland	Both*
		CS.4	Manage wet grassland for biodiversity	All	Wet grassland	Both*
		CS.5	Provide den boxes for pine martens	Operation	Forestry	Both

		CS.6	Plant small woodlands away from turbines	All	Forestry, Grassland*	Both
	Managing forestry for biodiversity	FM.1	Define felling timings	All	Commercial forestry	Both*
		FM.2	Maintain bat buffers with distance of treelines from turbines	All	Forestry	Both
		FM.3	Protect sensitive habitats during felling/maintenance works	All	Forestry	Both
		FM.4	Native tree planting on road verges	All	Forestry	Both*
	Manage existing water features in a biodiversity-friendly manner	WM.1	Protect riparian zones	All	Waterbodies & Watercourses	Both
		WM.2	Set up exclusion zones around watercourses	All	Waterbodies & Watercourses	Wholly-owned
		WM.3	Remove encroachment of conifer saplings from riparian corridors	Construction, Operation, Decommissioning	Waterbodies & Watercourses	Both
	Implement Nature-Based solutions around the site	NM.1		All	All	Both
	Curtailment	CU.1	Raise the cut-in speed	Operation	All	Both
		CU.2	Blanket curtailment	Operation	All	Both
		CU.3	Smart curtailment	Operation	All	Both
Restoration	Repowering/Extension of life opportunities	RE		Decommissioning	All	Both
	Peatland restoration	PR.1	Raise the water table to rewet peat	Planning, Operation, Decommissioning	Peatland	Wholly-owned
		PR.2	Fill or block drainage ditches	All	Peatland*	Both*
		PR.3	Plant peatland mosses/ <i>Sphagnum</i>	All	Peatland	Both
		PR.4	Forest to bog restoration	All	Peatland*	Both*
		PR.5	Remove conifer saplings and scrub encroachment on peatland	All	Peatland*	Both*
		PR.6	Implement deer management	All	Peatland	Both*
		PR.7	Bracken removal and gorse control on peatlands	All	Peatland	Both*
	Restore waterbody habitats	WR.1	Restore rivers for biodiversity	All	Rivers	Both*
Habitat Creation	Compensatory areas	CA.1	Cover crops for birds of prey/Hen Harrier	All	Agriculture, Forestry	Both*
		CA.2	Barn owl/Kestrel nest boxes	Operation	Agriculture	Both*

	Artificial pond creation	PA		All	Agriculture, Forestry	Both*
Biodiversity awareness (external)		BE.1	Community/Visitor iNaturalist identifications	Operation	All	Both*
		BE.2	Community Walk and Talks	Operation	All	Both*
		BE.3	Community co-design of biodiversity signage on site to inform visitors of biodiversity present	All	All	Both*
Invasive species management		IS.1	Conduct risk assessments for invasive species.	All	All	Both
		IS.2	Report the presence of invasive species	All	All	Both
		IS.3	Avoid the spread of non-native species during construction and maintenance	Construction, Operation, Decommissioning	All	Both
		IS.4	Ensure the correct removal methods for invasive plant species are followed	All	All	Both*
		IS.5	Identify and map the locations of invasive species	All	All	Both
		IS.6	Ensure staff working on site are aware of invasive species and are aware of how to identify and report invasive species should they be present on the site.	All	All	Both
Biodiversity monitoring		BM.1	Biodiversity Net Gain Monitoring	All	All	Both*

Example Site-Specific Windfarm Biodiversity Action Plan



Nature+Energy

Nature+Energy

November 2025

This Biodiversity Action Plan comprises an output of the Nature+Energy Project, funded by Taighde Éireann - Research Ireland (12/RC/2302_P2), industry partners (NTR Foundation, ESB, SSE Renewables, Asper Investment Management, Ecopower, Energia, EnergyPro, Greencoat Renewables, Ørsted) and MaREI, the Research Ireland Research Centre for Energy, Climate and Marine Research and Innovation.

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Policy and socio-cultural context

In 2019, Ireland declared a climate and biodiversity emergency and acknowledged the importance of healthy ecosystems for climate mitigation and adaptation measures. Ireland's most recent suite of County Development Plans seek to incorporate national and regional policy objectives and strategic outcomes (from the National Planning Framework and the relevant Regional Spatial and Economic Strategy) into local level objectives. The **recommended actions** in this Biodiversity Action Plan (BAP) for Nature+Energy Windfarm intentionally complement the national, regional and local level strategic objectives to protect, restore and enhance biodiversity at a site-specific level. These recommended actions are embedded in local community actions and initiatives and local socio-cultural contexts, in line with the recognition by Ireland's 4th National Biodiversity Action Plan 2023-2030 that "landowners, farmers, fishers, foresters and local communities are in many aspects the most important players in biodiversity issues". For example, the Nature+Energy BAP directly supports several of the Biodiversity Actions in the action-based Galway County Heritage & Biodiversity Plan 2024-2030, such as enabling the development of local biodiversity action plans in line with the All-Ireland Pollinator Plan 2021-2025 and the National Biodiversity Action Plan 2023-2030 and promoting best practice on how to conserve and restore biodiversity in County Galway. The proposed appointment of two Community Biodiversity Liaison Officers to support the development and delivery of biodiversity, education, conservation and restoration projects opens up an opportunity to connect the Nature+Energy BAP to local communities and other biodiversity initiatives in the county, such as through the proposed annual Biodiversity Community Engagement events. The recommended action in the Nature+Energy BAP to monitor and report the spread of invasive species, and to create and monitor marsh fritillary butterfly habitat, contributes to the Galway County Heritage & Biodiversity Plan 2024-2030 Biodiversity Objective to enhance opportunities for informed decisions through data collection. The county-level Biodiversity Action to support the conservation and restoration of wetland (including peatlands under coniferous forestry) is supported by the Nature+Energy BAP's recommended actions to fill or block drainage ditches on peatland habitats and to consider a forestry to peatland restoration project.

At a broader level, this BAP contributes to realising Ireland's national climate objective, which requires the State to "pursue and achieve, by no later than the end of 2050, the transition to a climate-resilient, biodiversity-rich, environmentally sustainable and climate-neutral economy", as reflected in the Vision Statement of the Galway County Council Local Authority Climate Action Plan 2024-2029. Galway County's Climate Action Plan highlights the interconnected nature of the climate and biodiversity crises and identifies as a strategic goal the critical need "to protect, restore, and enhance biodiversity and natural habitats in County Galway in order to increase the resilience of natural and human systems to climate change".

Similarly, the Galway County Local Economic and Community Plan 2024-2030 links the “protection, conservation and enhancement of natural heritage and biodiversity throughout the County” to the high-level goal of a “sustainable and climate resilient county” and notes that public consultation on the plan highlighted the protection and enhancement of biodiversity as a priority.¹ The Galway County Development Plan (CDP) 2022-2028 explicitly recognises the importance of protecting and enhancing biodiversity in the achievement of sustainable development, including the “biological diversity that occurs in the ordinary landscapes of the County”. The recommended action in this Biodiversity Action Plan to maintain bat buffers around turbines links directly to the CDP’s Natural Heritage and Biodiversity Objective 9 which seeks to protect bats and their roosts, feeding areas, flight paths and commuting routes.

The flourishing of biodiversity does not happen in a vacuum. While biodiversity monitoring and datasets provide essential information for biodiversity restoration and enhancement, personal narratives and lived experiences bring meaning and connection. Understanding biodiversity and cultural heritage through local community insights reflects the interconnections between nature and people through the richness of our natural and cultural heritage, in line with the objectives of the National Planning Framework. Stories and names that reflect traditions, language, and ecological knowledge help preserve both cultural identity and environmental awareness, while deepening engagement and fostering a sense of stewardship.

The EU’s Nature Restoration Law 2024 marks a transformative shift in the policy context and rhetoric from halting net biodiversity loss to nature restoration. Transformations require leaps of imagination that force us to think beyond our existing paradigms. They need space, at all levels, for imagining other possible worlds in the making and for imagining a world in which a multitude of possible worlds can fit. Understanding biodiversity and cultural heritage through local stories and names, is one way of creating space for a different starting point for conversations about the biodiversity, climate and pollution crises. From this starting point of collective human-nature connections, humans are framed as participants in an interdependent web of life, rather than protectors or destroyers who are in charge of and control natural resources. These names and stories have the creative capacity to open up locally rooted possibilities for worlds to emerge where humans and more-than-human species exist and flourish together, by bringing our attention to human-nature relationships of mutual care.

Site Overview

There were 24 habitat types (Fossitt Level 3) recorded across the site. Woodland was the most dominant habitat type on the site (80%), with conifer plantation (WD4) having the largest presence across the site covering 62% of the site area, with other types of woodland habitats such as recently-felled woodland (WS5) (9%), immature woodland (WS2) (4%) and scrub (WS1) (3%) also contributing to the site. Peatland habitats were also present and covered approximately 9% of the total site area, with cutover bog (PB4) being the most prevalent peatland type on the site (7%). Areas of heath and dense bracken made up approximately 4% of the total site area, with wet heath (HH3) contributing the majority of this habitat type, with small patches of dense bracken (HD1) and dry siliceous heath (HH1) also present. Cultivated and build land (4% of total site area) described much of the windfarm infrastructure or areas that were most likely to have been created/altered by the development of the windfarm. This includes areas of buildings and artificial surfaces (ED3), such as the substation (1% of total area), and areas of spoil and bare ground (ED2), which was used to describe most of the roadways throughout the site (2% of total area). Other habitat types in this category included areas of exposed sand and gravel (ED1) and areas of recolonising bare ground (ED3) which both made up <1% of the total site area. Throughout the site there were numerous water features and watercourses, which combined cover approximately 2% of the total area of the site. The largest of these were the acid oligotrophic lakes (FL2) but also present were dystrophic lakes (FL1), eroding/upland rivers (FW1) and areas of reed and large sedge swamps (FS1). There were several small patches of grassland throughout the site, covering <1% of the total site area, and found mostly near the site boundaries.



Table 1. Table of all the habitat types identified at the Nature+Energy Windfarm in 2024 using the Fossitt (2000) Guide to Habitats in Ireland. The area of each habitat is displayed in metres squared, in hectares, and as a percentage of the total area of the site.

Ecosystem type	Fossitt Code	Habitat Name	Area (m²)	Area (Ha)	Area (%)
Cultivated and built land	ED3	Buildings and artificial surfaces			1.27
	ED1	Exposed sand and gravel			0.11
	ED2	Spoil and bare ground			2.28
	ED3	Recolonising bare ground			0.24
	Total:				3.90
Freshwater	FL1	Dystrophic lakes			0.07
	FL2	Acid oligotrophic lakes			1.65
	FS1	Reed and large sedge swamps			0.06
	FW1	Eroding/upland rivers			0.16
	Total:				1.94
Grassland and marsh	GS2	Dry meadows and grassy verges			0.14
	GS3	Dry-humid acid grassland			0.02
	Total:				0.16
Heath and dense bracken	HD1	Dense bracken			0.33
	HH1	Dry siliceous heath			0.15
	HH3	Wet heath			3.95
	Total:				4.43
Peatlands	PB3	Lowland blanket bog			0.04
	PB4	Cutover bog			6.72
	PF1	Rich fen and flush			0.46
	PF2	Poor fen and flush			1.99
	Total:				9.21
Woodland and scrub	WD1	(Mixed) broadleaved woodland			0.19
	WD2	Mixed broadleaved/conifer woodland			0.73
	WD3	(Mixed) Conifer plantation			1.70
	WD4	Conifer plantation			61.95
	WS1	Scrub			3.00
	WS2	Immature woodland			3.71
	WS5	Recently-felled woodland			9.08
	Total:				80.35
Total:					100

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Species

There were many species identified as present on the site – some identified through Nature+Energy (N+E) during onsite data collection, and some from records submitted to the National Biodiversity Data Centre (NBDC).

From the NBDC records, mammal species recorded on the site included Eurasian badger (*Meles meles*), hedgehog (*Erinaceus europaeus*), European otter (*Lutra lutra*), and pine marten (*Martes martes*). The European Union's Habitat's Directive (Council Directive 92/43/EEC, 1992) covers both the European otter (Annex II) and the pine marten (Annex V). Many Irish mammal species are also protected under the Irish Wildlife Acts. During the surveys onsite, there was also evidence of deer, likely red deer (*Cervus elaphus*).

Some of the bird species that have been recorded on the site from the NBDC records included kestrel (*Falco tinnunculus* – red conservation status), red grouse (*Lagopus lagopus hibernicus* – red conservation status), snipe (*Gallinago gallinago* – red conservation status), golden plover (*Pluvialis apricaria* – red conservation status), and woodcock (*Scolopax rusticola* – red conservation status).

There was only one record of an invasive species within the NBDC records, the New Zealand Flatworm (*Arthurdendyus triangulatus*), which is a high impact invasive. While there were no identifications made on site, there are records of invasive plants such as Japanese knotweed (*Fallopia japonica*) and Rhododendron (*Rhododendron ponticum*) in surrounding areas.

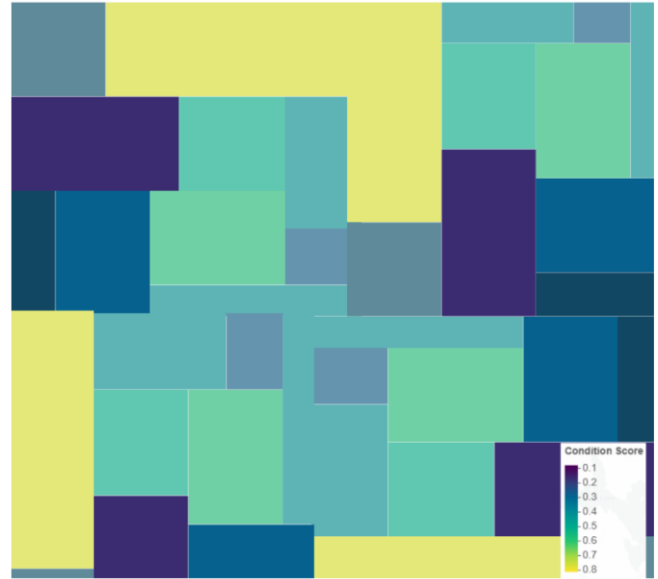


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Habitat condition

Through the process of ecosystem accounting, the condition of the habitats on site, in terms of their potential to provide resources for biodiversity was calculated. Condition scores are calculated between zero and one, where scores closer to zero indicate less potential to support biodiversity, and scores closer to one indicate a greater potential to support biodiversity. Habitat condition was calculated per habitat type but also per land parcel on the site. This allows us to identify areas that are weak in their potential for biodiversity, which may require greater management intervention or restoration, and areas where there is already likely good support for biodiversity, and this should be maintained and protected.

Nature+Energy’s Windfarm’s habitat’s had condition scores ranging from 0.08-0.81, with an average condition score of 0.37. Rich fen and flush (PF1) habitat had the highest average condition score (0.81), while lowland blanket bog (PB3) had the lowest average condition score (0.08).



Condition score map of Example Windfarm. Condition scores are shown on a colour gradient, going from lowest (darkest colour) to highest condition score (lightest colour). Condition score is an indicator of the habitat’s ability to provide space and resources for biodiversity.

Habitats and Recommended actions

We have provided a list of recommended actions for each habitat type found on the site below, following assessment of the habitat condition and site visits. The top actions that are recommended to be implemented site wide are outlined in Table 2, and the top three habitat-specific actions recommended are outlined in Table 3. Further details on these actions are outlined below in the relevant section.

Table 2. Top recommended actions that can be implemented across the whole site. Upward arrows indicate a positive indicator supporting taking this biodiversity action. The policy included is the Galway County Heritage and Biodiversity Action Plan 2024-2030 (CBAP).

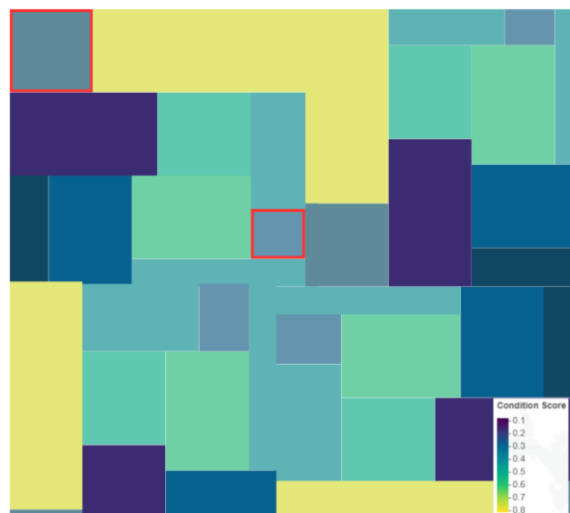
	Invasive species monitoring	Eliminate pesticide/herbicide use
Extent	Whole-site	Whole-site
Biodiversity benefit	High ↑	High ↑
Cost	Medium	Medium
Effort	Low ↑	Low ↑
Policy	CBAP ↑	

Table 3. Top three recommended actions for specific habitat types. Upward arrows indicate a positive indicator supporting taking this biodiversity action. The policy included is the Galway County Heritage and Biodiversity Action Plan 2024-2030 (CBAP).

	Marsh Frithillary habitat creation	Rush management	Forest to bog restoration
Extent	Peatlands	Woodland/Peatland	Cutover bog
Biodiversity benefit	High ↑	High ↑	High ↑
Cost	Medium	Medium	High
Effort	Medium	Medium	High
Policy	CBAP ↑		CBAP ↑

Woodland and scrub

WD1 Broadleaf woodland



Broadleaf woodland (WD1) are woodland areas where there is 75-100% cover of broadleaf trees, with less than 25% conifer cover. On the site, these areas occur in thin strips adjacent to immature woodland and fen habitats. This area had a condition score ranging from 0.33-0.49, with a mean score of 0.43. Broadleaf woodlands comprise important areas for many bird species, invertebrates, and mammals, providing shelter and food resources. The main risks identified to this habitat was the high tree cover density. When trees are tightly packed into a narrow space, and the canopy cover is high it can reduce the availability of light for other plant species that use the lower canopy layers, and it can also increase the risk of spread of disease from which some broadleaf species can be particularly vulnerable.

Recommended actions

- Eliminate pesticide and herbicide use, where possible
- Monitor and report the spread of invasive species
- Allow the forest to naturally thin out overtime
- Leave stacks of deadwood for invertebrates and small mammals to use as a microhabitat
- Provide den boxes for pine martens

Leave stacks of deadwood

Deadwood, which is woody material which has fallen in a forest, can be stacked and left in-situ to provide microhabitats for wildlife. Many species are associated with deadwood, in particular there are many mosses, fungi, and invertebrates that will rapidly colonise and utilise this microhabitat. In addition, the decay of these stacks can help with nutrient cycling in the soil. Deadwood occurs naturally but can be stacked to create different layers at different stages of decomposition, which provides different conditions for different species. For some deadwood that fell naturally, a minimal intervention approach is preferred to stacking, as it allows the colonisation and decay approaches to occur most naturally.



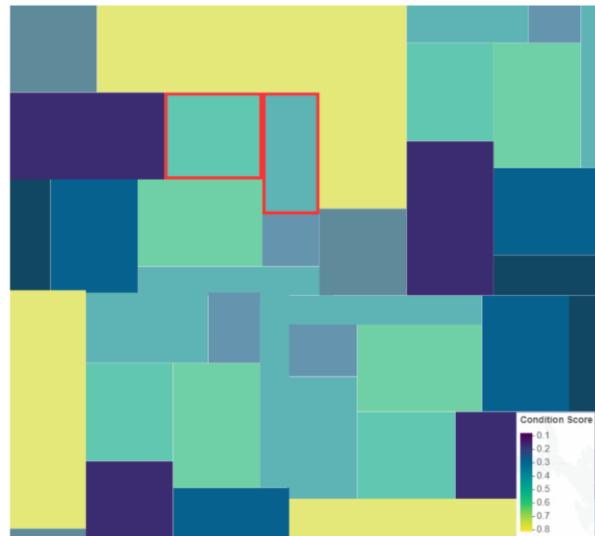
Provide den boxes for pine marten

Pine martens (*Martes martes*) are a native Irish mammal species, which have suffered population declines throughout the 20th century, but more recently numbers have been increasing. If the above areas of woodland will remain in place and not be felled for forestry or operational purposes and are located away from areas of disturbance, pine marten den boxes could be installed to increase the number of suitable breeding sites. Pine martens typically raise their young at height in old trees. Den boxes can provide pine martens with a safe space to raise their young kits. Boxes should be placed in older trees which have branches below them. Trees with coarser bark should also be used to provide easier access for pine martens. Pine marten boxes can be bought or made. Local community groups (e.g., Men's Sheds) could be involved in making the boxes or it can be staff activity. [The Vincent Wildlife Trust](#) provides comprehensive information on installing pine marten den boxes.



Pine marten den box © Vincent Wildlife Trust

WD2 Mixed broadleaf/Conifer woodland

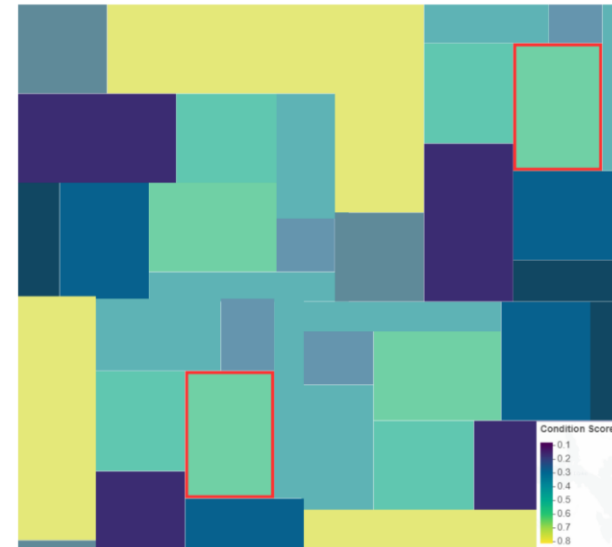


Mixed broadleaf/conifer woodlands (WD2) are woodlands which have a mix of both broadleaf species and conifer species, in which both types have a minimum coverage of 25% and a maximum coverage of 75%. On the site they primarily occurred alongside some road verges. The condition score of this habitat type ranged from 0.36-0.44, with a mean condition score of 0.38. The main risk identified was the high density of the tree cover and canopy. The main risks identified to this habitat was the high tree cover density. When trees are tightly packed into a narrow space, and the canopy cover is high it can reduce the availability of light for other plant species that use the lower canopy layers, and it can also increase the risk of spread of disease from which some broadleaf species can be particularly vulnerable.

Recommended actions

- Eliminate pesticide and herbicide use, where possible
- Monitor and report the spread of invasive species
- Allow the forest to naturally thin out overtime
- Leave stacks of deadwood for invertebrates and small mammals to use as a microhabitat
- Provide den boxes for pine martens

WD3 (Mixed) Conifer woodland

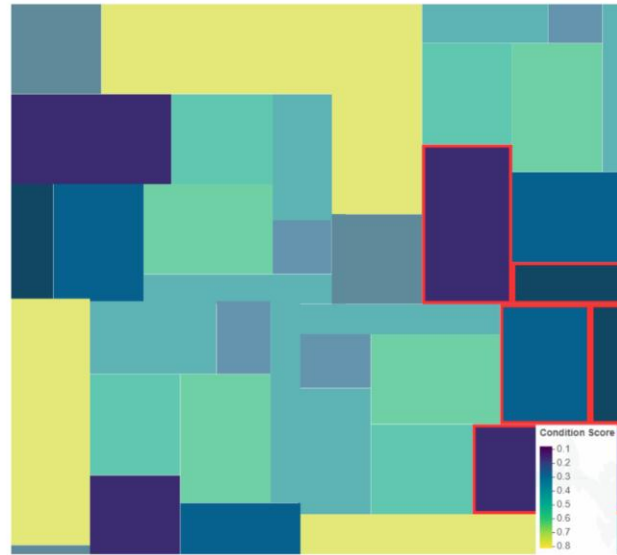


Mixed conifer woodland (WD3) is categorised by woodland areas with at least 75-100% conifers but there are still some broadleaf species present (<25%). They are termed mixed due to the presence of several different types of conifer trees located within the stand. They had a condition score ranging from 0.31-0.33, with an average score of 0.38. The main risks identified to this habitat were a high tree cover density and poor drainage, in that for peat soil the soil layers were very dry. This is likely occurring due to the high density of conifers planted on the peat soils.

Recommended actions

- Eliminate pesticide and herbicide use, where possible
- Monitor and report the spread of invasive species
- Allow the forest to naturally thin out overtime
- Leave stacks of deadwood for invertebrates and small mammals to use as a microhabitat
- Explore options for blocking drainage ditches within this habitat

WD4 Conifer plantation



Conifer plantation (WD4) was the dominant habitat type on the site. Conifer plantation differs from the other woodland on the site in that it was primarily planted for commercial timber production. They have a characteristic planting structure, with the trees planted in regular rows in angular blocks. The primary species in the conifer plantations was Sitka spruce (*Picea sitchensis*) and lodgepole pine (*Pinus contorta*), with few other species identified within the plantations. Condition scores ranged from 0.21-0.61, with an average condition score of 0.39. The main risks that were impacting the condition scores for biodiversity were the high tree cover density, which limits light to understory layers preventing vegetation growth, and the lack of species richness within the stands, which limits the biodiversity that can utilise these areas. Much of this area is outside the operational boundary of the windfarm and under the control of the forestry operator, therefore there are limits to what can be done to improve the condition scores of these areas. One of the main actions that is recommended is to maintain bat buffers in terms of distance of the treelines from turbines. If the areas of conifer plantation will remain on site and not be felled, pine marten boxes could be installed.

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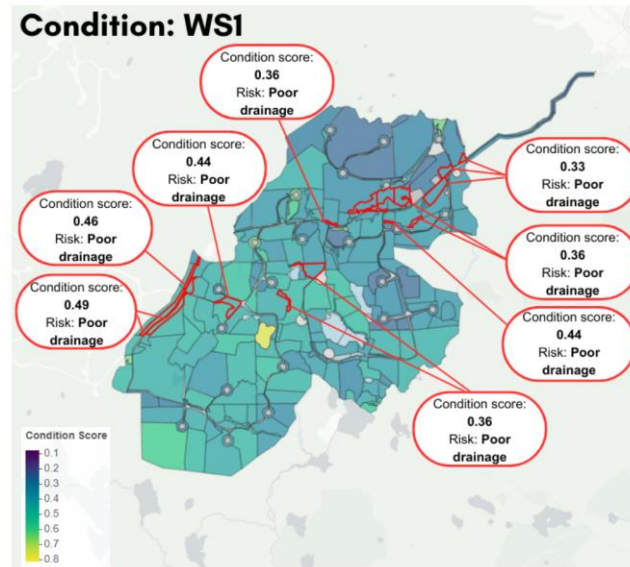
Maintain bat buffers

Most Irish bat species preferentially fly along linear features, such as treelines or hedgerows. Felling creates new treelines which may increase bat activity (Kirkpatrick et al., 2017). Many windfarms currently follow the *NatureScot Guidance* of a 50 metre buffer zone between the blade tip and the nearest woodland. This should be maintained throughout the lifespan of the windfarm. This involves removing any conifer encroachment periodically within the 50-metre buffer of the turbine. This should be conducted on a yearly basis to ensure that conifer saplings are removed at the earliest opportunity and while they are still relatively small in size.



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WS1 Scrub

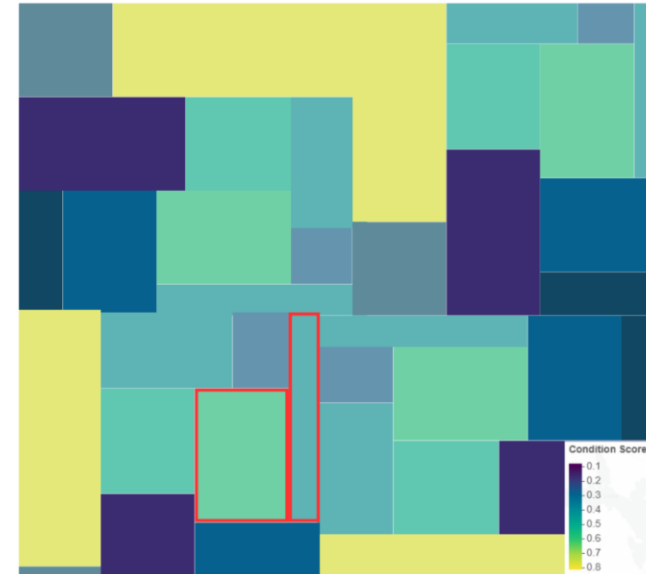


Scrub (WS1) occurred throughout the site. The main components of the scrub habitats were gorse (*Ulex europaeus*) and bramble (*Rubus fruticosus*). Scrub can provide shelter and food resources for invertebrates, mammals and small bird species, however, on peatland soils it can dry out the peat which can affect the unique peatland biodiversity and its ability to sequester carbon. The scrub habitats on the site ranged in condition score from 0.33-0.49. The main risk identified was poor drainage, likely caused by the scrub vegetation draining water from the peatland soils, affecting the surrounding vegetation.

Recommended actions

- Eliminate pesticide and herbicide use, where possible
- Monitor and report the spread of invasive species
- Implement a grazing management plan. Grazing can help reduce scrub encroachment and maintain the vegetation structure. Extensive grazing at low stocking rates is recommended.
- Designate the area for wildlife
- Monitor and report the spread of invasive species
- Identify and block any existing drains that may be contributing to the poor drainage in these areas

WS2 Immature woodland

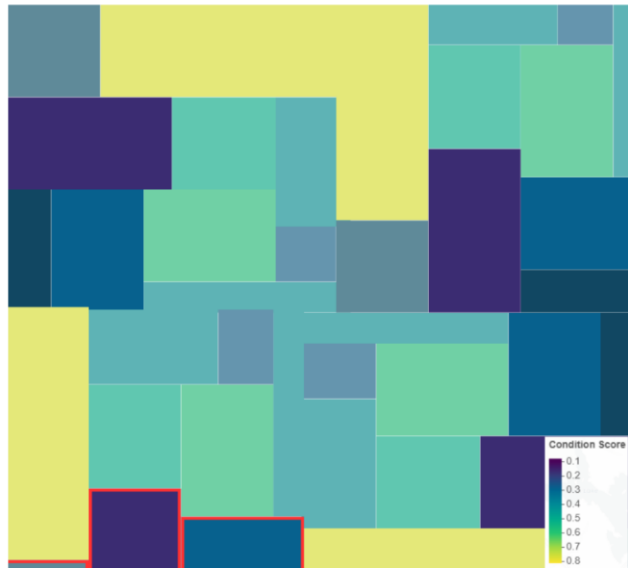


Immature woodland includes areas that are dominated by young trees that have not yet reached the height threshold for the woodland categories in Fossitt (2000) *A Guide to Habitats in Ireland*, where the threshold height is defined as 5 metres. It is likely these are areas of previous conifer plantation that have since been replanted with young saplings. The condition score of this habitat ranged from 0.28-0.44, with a mean condition score of 0.39. The main risk identified was the lack of presence of deadwood, an important microhabitat in woodlands for invertebrates, fungi and mosses.

Recommended actions

- Eliminate pesticide and herbicide use, where possible
- Monitor and report the spread of invasive species
- Leave stacks of deadwood for invertebrates and small mammals to use as a microhabitat

WS5 Recently-felled woodland



Recently-felled woodland (WS5) are areas which have been clear-felled but not replanted and have not yet transitioned into a different habitat. The condition scores for this habitat ranged from 0.21-0.58, with an average condition score of 0.35. The main threats identified for this habitat type were poor drainage and low species richness. Poor drainage is present likely due to the previous habitat of conifer plantations which often creates drains on peatland soils. The low species richness is especially present in more recently felled areas, where there has not been much time for recolonisation, therefore there is little biodiversity currently present in these areas.

Recommended actions

- Monitor and report the spread of invasive species. Recently-felled areas have a higher risk of invasive species establishment.
- Eliminate pesticide and herbicide use, where possible
- Block any remaining drains present from conifer plantations
- If landowner agrees, consider options to collaborate on a forestry to peatland restoration project

Forest to bog restoration

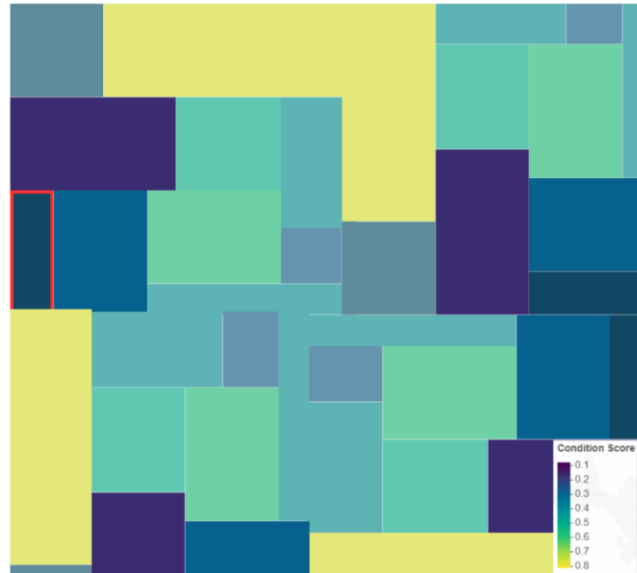
If agreed with the landowner/operator, consider that recently-felled areas be transitioned and restored to peatland habitats. This would also need to be in agreement with the felling and replanting licence under the Forestry Act 2014. The goal should be to reintroduce the peatland's natural hydrological function and restore the natural typology of the peat surface to promote the recovery of vegetation communities. [NatureScot](#) provides best practice for forest to bog restoration.



An example of forest to bog restoration on an Irish windfarm. The conifer plantation has been cleared back to connect the restored peatland with an existing area of peatland in the background. Note the bright green and red mosses are *Sphagnum* mosses, which are an indicator of an active peatland system.

Peatlands

PB3 Lowland blanket bog



There was only one area of lowland blanket bog (PB3) on the site along the boundary to the north. It had a condition score of 0.08 and the main risks identified were bare peat and encroachment from scrub and the nearby conifer plantations. Bare peat can be problematic for bog habitats as it indicates that the bog may not be active and could risk emitting carbon that is usually stored within the peat. Peatland habitats are important habitats for biodiversity supporting a unique flora and fauna, such as *Sphagnum* mosses. *Sphagnum* is a key component of peatland habitats and contribute to healthy peatland functioning and carbon sequestration.

Recommended actions

- Designate areas for wildlife
- Monitor and remove conifer scrub and encroachment
- Monitor and report the spread of invasive species
- Eliminate pesticide and herbicide use near these sensitive habitats
- Consider blocking any drains near this habitat to increase wetness
- Limit and avoid, where possible, trampling by machinery, traffic, or people

Monitor and remove conifer and scrub encroachment on peatland

Due to the nearby plantations, scrub and conifer encroachment are often prevalent. It can be important to monitor for and remove scrub and conifer encroachment as they can draw water out of the peatlands and cause them to dry out, and in addition they can also share out peatland vegetation. The removal of large areas of scrub should be done with caution, however, due to the risk of causing peatland instability, and only outside of the March-August bird breeding season. This is a continual practice, where encroachment will need to be monitored, and actions implemented continuously throughout the lifetime of the windfarm. *NatureScot* provides best practices on scrub and conifer removal on peatlands and it may be that a combination of techniques needs to be used. Landowners should be consulted prior to implementing this action. Conducting monitoring and removal of conifer and scrub encroachment should be seen as beneficial to all parties, therefore we encourage conversations with landowners, where possible, about the feasibility of implementing this action and who would be responsible for it.



Fill or block drainage ditches

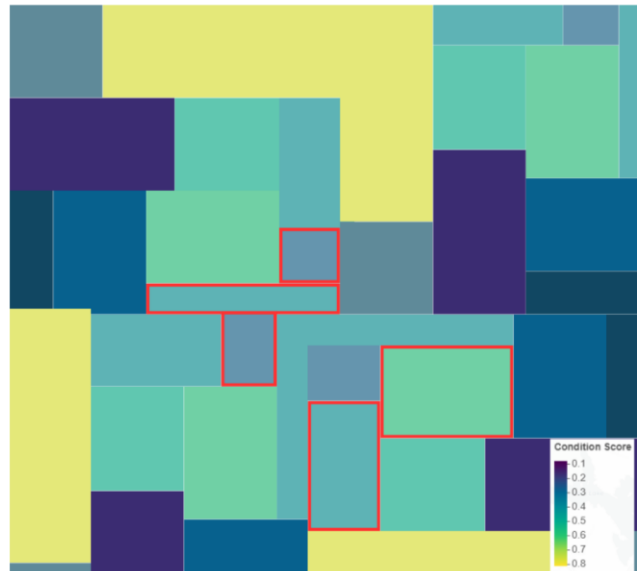
Drains and ditches extract water from the surrounding peatland and peatland vegetation cannot establish in ditches that contain deep water or where there are large volumes of flowing water. Prioritising locations to block is important to maximise the efficiency of using dams and an engineer with local knowledge should be consulted regarding this. Drains can be blocked with dams to create shallow pools which encourage peatland plants to grow and eventually lead to peatland deposition. Peat dams are often used to block artificial drains when there is still sufficient peat remaining at the bottom of the drain. *NatureScot* provides detailed guidance on this method. Landowners and operators should be consulted prior to implementing this action, as it may impact other activities (e.g., silviculture).



Peatland drain blocking examples

© NatureScot

PB4 Cutover bog



Cutover bog (PB4) habitats are peatland habitats which have been cut or disturbed. There remain some remnants of peatland vegetation (e.g., *Sphagnum*) but there are usually large areas of bare and drained peat, and the area of these depends on the time since disturbance. The condition of the cutover bogs on the windfarm ranged from 0.21 to 0.65 with an average condition score of 0.32. The main risks to the condition of this habitat that were identified was bare peat and poor coverage of *Sphagnum*.

Recommended actions

- Fill or block drainage ditches
- Remove conifer and scrub encroachment
- Monitor and report invasive species
- Eliminate pesticide and herbicide use near these sensitive habitats
- Designate areas for wildlife
- Implement deer management
- Raise the water table to rewet the peatland surface
- Marsh fritillary habitat creation
- Manage rushes

Implement deer management

Sightings and signs of deer were recorded on the site during visits. There is evidence that the presence of deer within peatlands can be harmful of the long-term conservation of peatland plant diversity, especially if they occur in high densities (Pellerin et al., 2006). A deer management plan may be needed to manage deer populations at sustainable levels to reduce grazing and trampling on bogs, especially in areas that are undergoing restoration. This would be particularly important for areas of cutover bog that have bare peat as a risk as some of this can be caused by over-trampling. Deer management can include deer-proof fences or in some cases culling. Culling is subject to licence. Prior to implementing such management, it is best to monitor for signs of overgrazing and over-trampling at the relevant habitat patches on the site.



Signs of trampling and deer on the site.

Raise the water table to rewet the peatland surface

Raising the water table is a key process of peatland restoration and is often recommended for cutover bogs where they may have been previously drained. However, on the site, much of the cutover bog habitat occurs near the turbine hardstands or along the roadsides, thus it may not be best suited to rewetting as it may pose a risk to the infrastructure.



Sundews are a sign of a wet, healthy peatland habitat

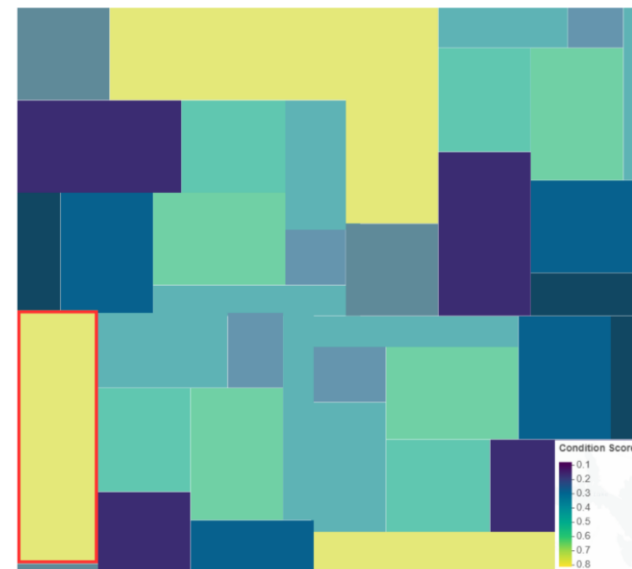
Manage rushes

Although native, rushes can infest a grassland and reduce the overall plant diversity. Rushes often infest poorly drained, compacted soils, which can be caused by over-trampling by livestock in the agricultural setting. On windfarms rush infestation can be caused by soil compaction during the reinstatement of soils during the construction phase. If the goal is to increase the grassland diversity, rushes may need to be managed throughout the windfarm's lifespan to reduce infestation. From September onwards, rushes can be topped. Check for the presence of ground-nesting birds prior to topping rushes. A first cut in late Autumn followed by a second cut weeks later can help reduce rush cover. Following topping, grazing can also be used to control it. Rush management is recommended to be included in a management plan for the windfarm, where rushes are managed on a two-year cycle.



Rush infestation around one of the turbine hardstands

PF1 Rich fen and flush

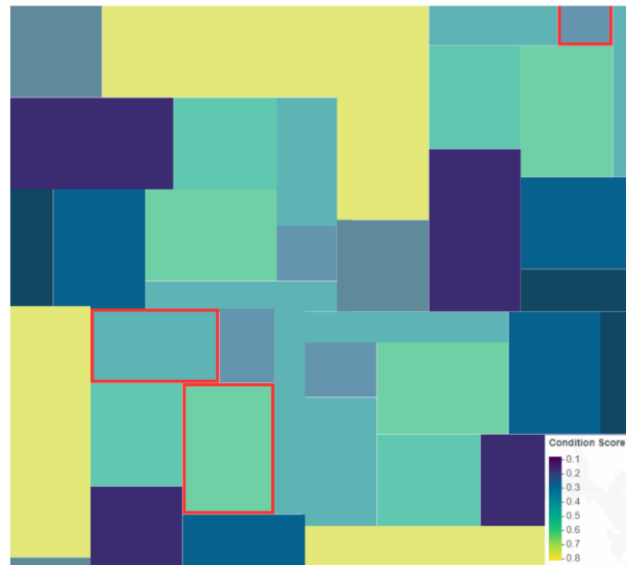


There was one large area on the site which was identified as rich fen and flush. Fen habitats are similar to peat bogs in that they also form peat however they are fed by groundwater. This habitat had a high condition score of 0.81 and the primary risk identified was scrub and conifer encroachment from the surrounding conifer plantations. Scrub and conifer encroachment can dry out fen habitats which can reduce the condition of the habitat for biodiversity. This habitat type is sensitive to input from excess nutrients from the surrounding land and other pollution sources. *NatureScot* has provided an extensive handbook on fens which provides useful guidance to maintaining fens: [The Fen Management Handbook](#).

Recommended actions

- Implement a grazing management plan. Grazing can help reduce scrub encroachment and maintain the vegetation structure. Extensive grazing at low stocking rates is recommended.
- Eliminate pesticide and herbicide use near these sensitive habitats
- Monitor and report the spread of invasive species
- Eliminate pesticide and herbicide use near these sensitive habitats
- Monitor and remove scrub encroachment, preferably by hand.
- Designate area for wildlife.
- Reduce foot traffic/machinery use on this area of the site.

PF2 Poor fen and flush



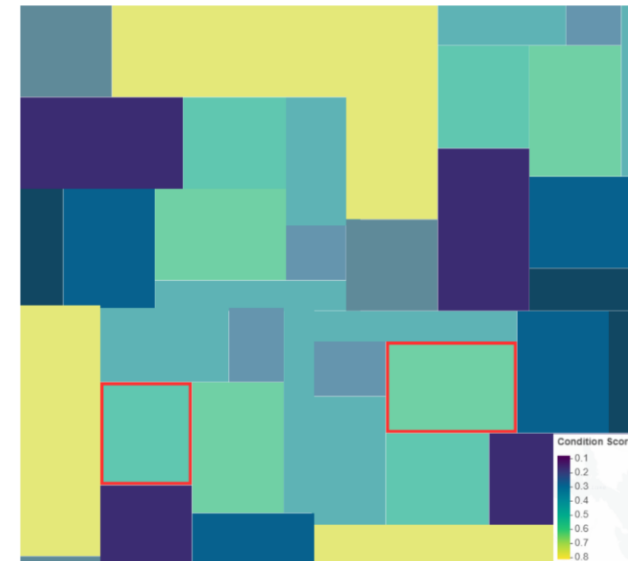
There were several areas across the site that were identified as poor fen and flush (PF2). The condition score of this habitat ranged from 0.44-0.57, with an average condition score of 0.52. The primary risk identified to this habitat was scrub encroachment from the surrounding areas. Scrub and conifer encroachment can dry out fen habitats which can reduce the condition of the habitat for biodiversity. *NatureScot* has a handbook for managing this type of habitat correctly: [The Fen Management Handbook](#).

Recommended actions

- Implement a grazing management plan. Grazing can help reduce scrub encroachment and maintain the vegetation structure. Extensive grazing at low stocking rates is recommended.
- Eliminate pesticide and herbicide use near these sensitive habitats
- Monitor and report the spread of invasive species
- Monitor and remove scrub encroachment, preferably by hand
- Designate area for wildlife
- Reduce foot traffic/machinery use on this area of the site

Heath and dense bracken

HD1 Dense bracken



Dense bracken (HD1) is usually characterised by areas that are dominated by bracken (*Pteridium aquilinum*). There were four areas of dense bracken on the site, with condition scores ranging from 0.34-0.42, and an average condition score of 0.38. Areas of this habitat type were primarily located along road verges, so possibly formed as a result of disturbance during the roads construction. The main risks identified to the condition of this habitat were scrub encroachment and areas of bare peat within the habitat. Dense bracken usually has low species richness as a habitat type.

Recommended actions

- Monitor and report the spread of invasive species
- Bracken strimming
- Fence off areas of dense bracken
- Reduce and, where possible, eliminate pesticide and herbicide use

Bracken management

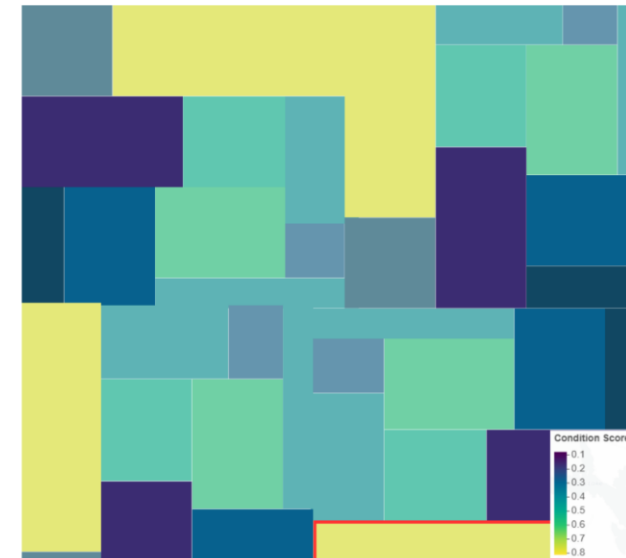
Bracken can spread quickly and encroach into surrounding areas, which can have negative impacts on surrounding peatland habitats. Monitor for encroachment of the bracken in bordering habitats and if necessary, cut (by hand strimming) the bracken to reduce spread.

Fence back the bracken along the boundaries of the habitat to reduce spread and ensure that visitors and site staff do not come into contact with the bracken as it can host ticks, and it increases the risk of the spread of Lyme's disease.



An example of an area of dense bracken © Cornfoot (2009)

HH1 Dry siliceous heath



There were two areas in the north of the site which were identified as dry siliceous heath (HH1). Both areas had a condition score of 0.65. Heath habitats have similar vegetation to peatlands, however, do not have wet peat soils. There are important habitats for biodiversity, including species such as the marsh fritillary butterfly (*Euphydryas aurinia*) and Irish hare (*Lepus timidus hibernicus*). The main risk identified to this habitat was scrub and conifer encroachment from the surrounding areas.

Recommended actions

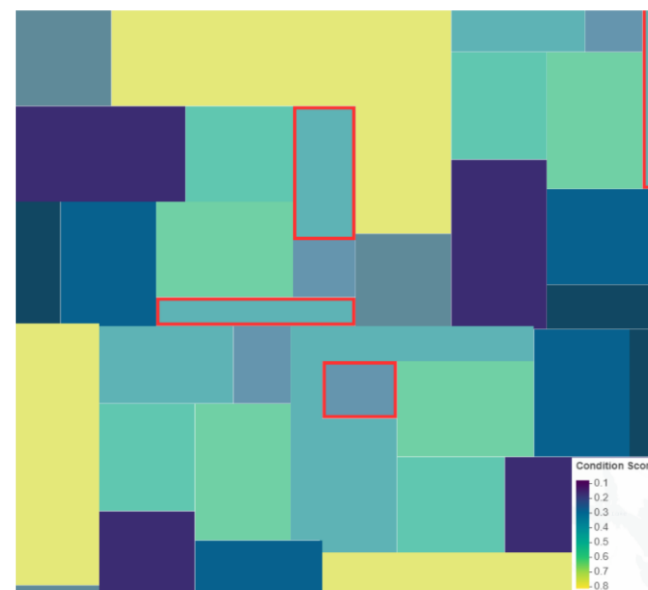
- Monitor and report the spread of invasive species
- Reduce and, where possible, eliminate pesticide and herbicide use
- Enforce a scrub management plan. Monitor scrub encroachment and remove patches of scrub
- Designate areas for wildlife
- Marsh fritillary habitat creation

Marsh fritillary habitat creation

The Marsh Fritillary butterfly (*Euphydryas aurinia*) is Ireland's only insect on Annex II of the Habitat's Directive (Council Directive 92/43/EEC, 1992). It is found across Ireland, and according to the [National Biodiversity Data Centre Database](#) it has been recorded in the surrounding areas of the windfarm. The marsh fritillary's main food source is Devil's Bit Scabious (*Succisa pratensis*) which flowers from July-September. If this plant is found among the heath, it may be an indicator of the presence of marsh fritillary butterfly. If present, monitoring should be done to record populations of the butterfly and monitor the status of the habitat over time. The NBDC has been running a [monitoring scheme](#) since 2015 and has provided advice on monitoring for this species and habitat. The NBDC also has a [comprehensive guide](#) on habitat management for the marsh fritillary.



HH3 Wet heath



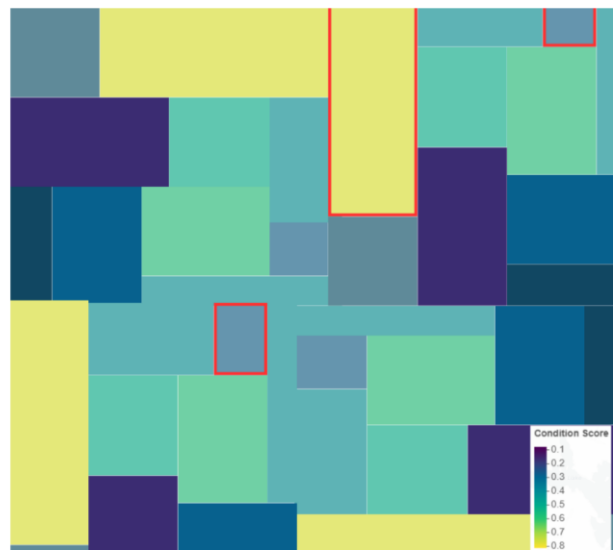
Wet heath (HH3) habitats are often characterised by waterlogged soils which have heather, grasses and sedges. They are important for biodiversity, especially invertebrates and amphibians. There were many areas of wet heath on the site with an average condition score of 0.41 and a range of 0.23 to 0.48. The main risk identified to this habitat was overgrazing, primarily from sheep.

Recommended actions

- Enforce a scrub management plan. Monitor scrub and conifer encroachment and remove patches of scrub.
- Reduce and, where possible, eliminate pesticide and herbicide use
- Monitor and report the spread of invasive species
- Designate areas for wildlife
- Marsh fritillary habitat creation
- Block drains occurring near the wet heath to ensure soils remain waterlogged throughout the year.
- Do not use pesticides/herbicides/fertilisers near this habitat as it can affect vegetation structure and growth.

Cultivated and built land

ED3 Recolonising bare ground



The condition of recolonising bare ground (ED3) habitats ranged from 0.38-0.67, with an average score of 0.51. The main risks identified to the condition of this habitat were scrub encroachment and the presence of large areas of bare soil, possibly caused by trampling from machinery.

Recommended actions

- Remove conifer and scrub encroachment
- Reduce and, where possible, eliminate pesticide and herbicide use
- Monitor and report the spread of invasive species
- Designate areas for wildlife

Monitor and remove conifer and scrub encroachment

Due to the nearby plantations, scrub and conifer encroachment are often prevalent. It can be important to monitor and remove scrub and conifer encroachment as they can prevent native, natural vegetation from recolonising these areas. While scrub can be beneficial for wildlife, given that the soils and surrounding areas are peatland, scrub can dry out these environs. Therefore, it is best to monitor scrub encroachment and where necessary remove it to prevent impact on peat soils and habitats. Conifer saplings from the nearby plantations should also be removed, where possible, as these species are non-native and can negatively impact peat soils. Removing conifers at the sapling stage is the easiest time, and limits damage to the peatland soils. The removal of large areas of scrub should be done with caution, however, due to the risk of causing peatland instability and should only be conducted outside of the March-August bird breeding season. Monitoring and removal are continual practices, where encroachment will need to be monitored, and the action continuously implemented throughout the lifetime of the windfarm. Landowners should be consulted prior to implementing this action. Conducting monitoring and removal of conifer and scrub encroachment should be seen as beneficial to all parties, therefore we encourage conversations with landowners, where possible, about the feasibility of implementing this action and who would be responsible for it.



Monitor and report the spread of invasive species

While no invasive species were identified during surveys, there are some present in the surrounding environments. The presence of large areas of bare soil can create space for invasive plant species to settle and grow, thus in particular these areas should be monitored to ensure that they do not become established within the site. The National Biodiversity Data Centre (NBDC) runs [Invasives.ie](https://invasives.ie) which provides updated, useful resources for the identification and management of invasive species. A map of the locations of *Rhododendron* identified on the site is included below.

Invasive species management

- Conduct a risk assessment for invasive species to identify the risk of spread and plan for management and removal methods (if approved under licence).
- Report the presence of invasive species through the [Invasives.ie](https://invasives.ie) portal.
- Avoid the spread of invasive species during maintenance works on site through education and awareness campaigns with staff.
- Regulation 49 of S.I. 477 outlines that removal of an invasive species requires a licence from the National Parks and Wildlife Service.

Designate areas for wildlife

Areas of bare ground usually arise from disturbance caused by the installation of wind turbines or their infrastructure. These areas may be left unused by other land uses and so can be left as areas of resources for wildlife. To be so, they should be managed in a biodiversity-friendly manner (no pesticides/herbicides, minimal disturbance, and following the actions recommended above). Signage can be erected to highlight that these areas are being managed in a positive way for wildlife.

Freshwater features

FL1 Dystrophic lakes

FL2 Acid oligotrophic lakes

FW1 Eroding/upland rivers

There were several freshwater features on the site, which are all important for biodiversity. Freshwater habitats can support a range of species, especially invertebrates. They also provide resources for many other species, such as mammals, and birds. Protecting the features themselves, and their surrounding areas is important to maintaining many ecosystem services of the local area.

Recommended actions

- Protect riparian zones
- Set up exclusion zones around watercourses
- Reduce and, where possible, eliminate pesticide and herbicide use
- Remove conifer encroachment from riparian areas
- Designate areas for wildlife



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Protect the riparian zones

The riparian zone is the transitional areas between the waterbody and the adjacent habitat. These areas often have high biodiversity potential, if managed and maintained correctly, and act as an important buffer to protect the watercourse from any agriculture/silviculture run-off or pollutants. It is recommended to work with the landowner to ensure this important habitat is maintained on the site.

Remove conifer encroachment from riparian areas

The encroachment of non-native conifer trees (e.g., Sitka spruce *Picea sitchensis*) into riparian zones can cause these habitats to dry out and can overtake native vegetation. The presence of conifer trees can also lead to increases in acidification and nutrients into these systems, harming aquatic biodiversity and water quality. The aim should be to maintain a buffer zone of at least 20 metres between the boundary of the conifer plantation and watercourses. Conifer encroachment will need continuous monitoring throughout the lifespan of the windfarm. This can be collaborative between the landowner/forestry operator and the windfarm operator and management.

Set up an exclusion zone around watercourses

Native vegetation surrounding waterbodies provide important ecosystem services, such as the provision of habitat, water quality, and carbon sequestration. Overgrazing or trampling from human activities and machinery can affect the ability of these areas to provide ecosystem services. If heavy grazing becomes an issue, fencing (that still allows wildlife to pass) can be placed around the waterbody, at a distance of 20-30 metres from the waterbody itself. If human activities are creating disturbance, signage and education around the importance of these areas can be shared, especially to those who may be conducting works around these areas. This can be aligned with designating the area for wildlife, which places emphasis that this area is being managed in a positive manner for wildlife.

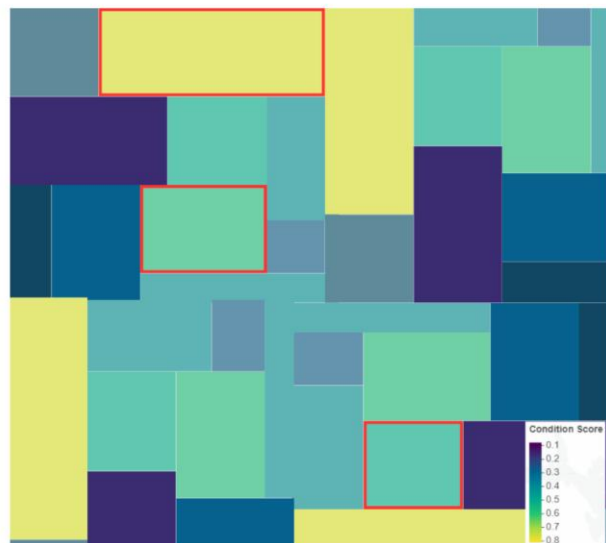


Conifer trees and encroachment growing in the riparian zone of a lake

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Grassland and marsh

GS2 Dry meadows and grassy verges

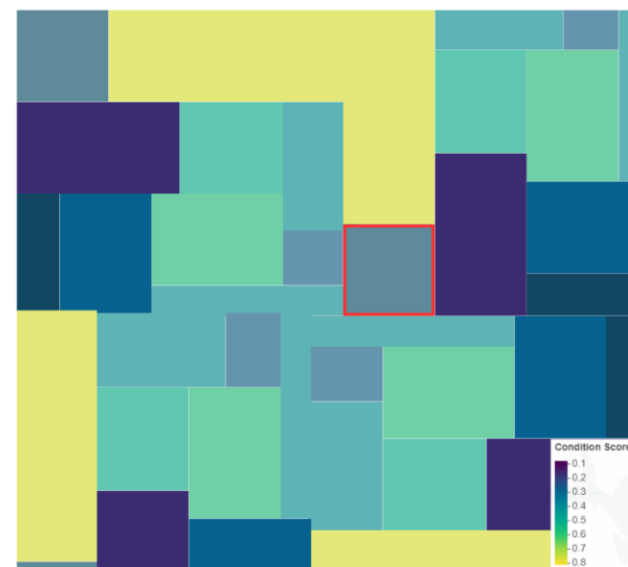


There were three small patches of dry meadows and grassy verge (GS2) habitats within the windfarm boundary. These habitats likely arose due to habitat changes during the installation of the wind turbine infrastructure or the site roadways. These processes can introduce heterogeneity into the site's landscape. Introduced heterogeneity is important for wildlife and especially important for providing resources for pollinators, small mammals, and small bird species. The habitat condition of these habitats ranged from 0.33-0.56, with an average condition score of 0.45. The main risks identified were scrub encroachment and overgrazing.

Recommended actions

- Remove scrub/conifer encroachment
- Monitor and report the spread of invasive species
- Reduce and, where possible, eliminate pesticide and herbicide use
- Manage livestock to prevent overgrazing. If possible, arranging livestock to rotate more frequently between habitat patches to reduce the pressure on areas that are overgrazed.

GS3 Dry-humid acid grassland



There was one area of dry-humid acid grassland (GS3) located on the site, with a condition score of 0.33. The main risk identified to this habitat was the presence of large areas of bare soil within the habitat.

Recommended actions

- Monitor and report the spread of invasive species
- Fence off with wildlife-friendly fencing and push out this fence to limit the grazing pressure on the grassland. This will limit overgrazing and trampling by sheep in the grassland.
- Designate the area for wildlife.
- Enforce a scrub management plan. Monitor scrub encroachment and remove patches of scrub.
- Manage the area in a positive way for pollinators
- Do not use herbicide/pesticide in this area or along adjoining road verges, where possible.

Manage grassland for pollinators

If these areas are beyond the boundary of the windfarm operator, it may be possible to collaborate with the landowner to see if the land management methods can be improved for biodiversity, and in some cases, this involve doing less intensive management. Leaving areas of long grasses and wildflowers provides nesting resources for bumblebees and food for many of Ireland's bee species. The [All-Ireland Pollinator Plan](#) has a lot of resources available for landowners and windfarm operators regarding managing land positively for pollinators.



An example of grassland on a windfarm

Appendix D – Supplementary Information for Chapter

D1. Acoustic recorder locations

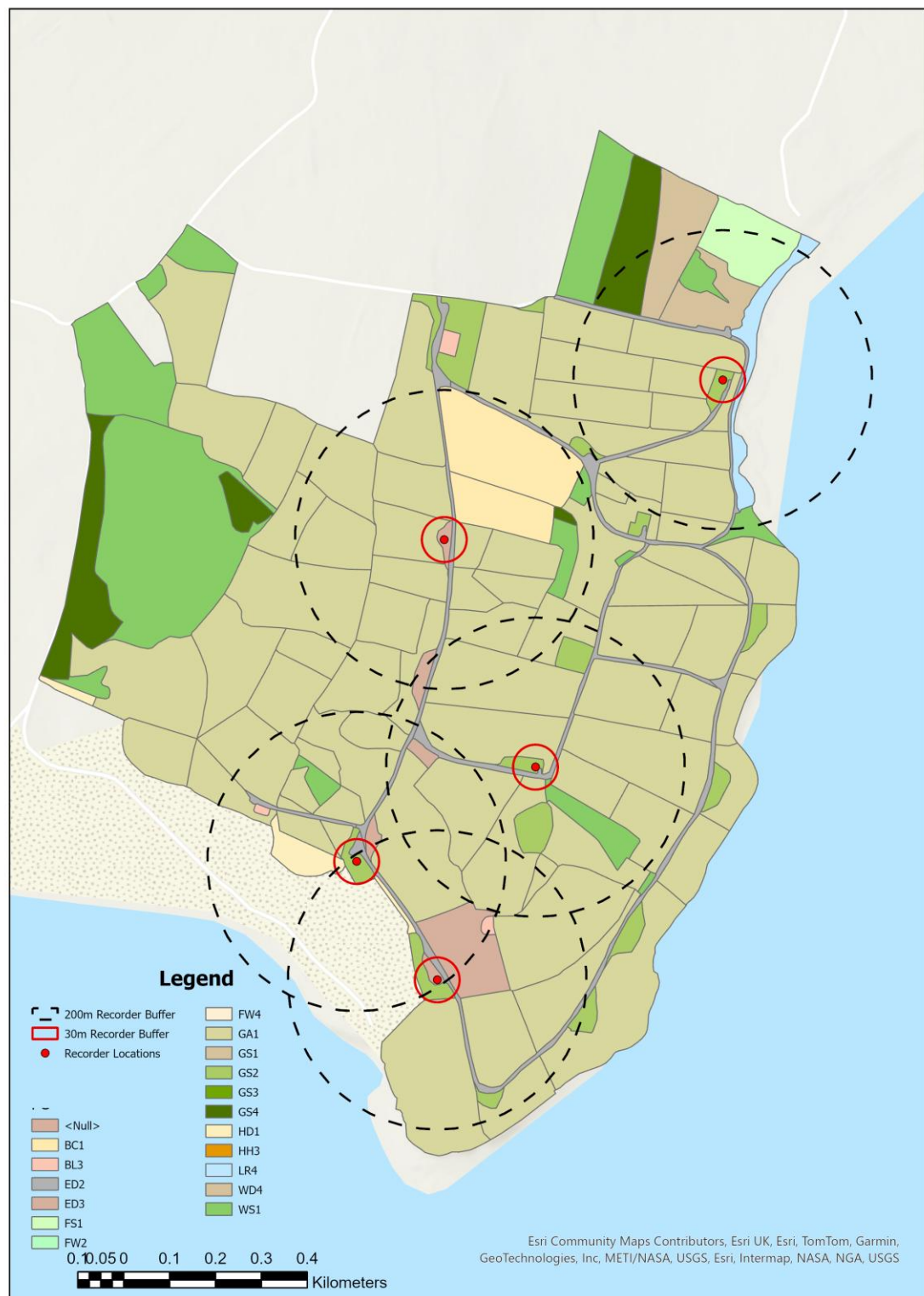


Figure D.1. Acoustic recorder locations on Carnsore windfarm. Recorder locations are shown as points, the 30 m spatial buffer is a red circle, and the dashed circle is the 200 m spatial scale.

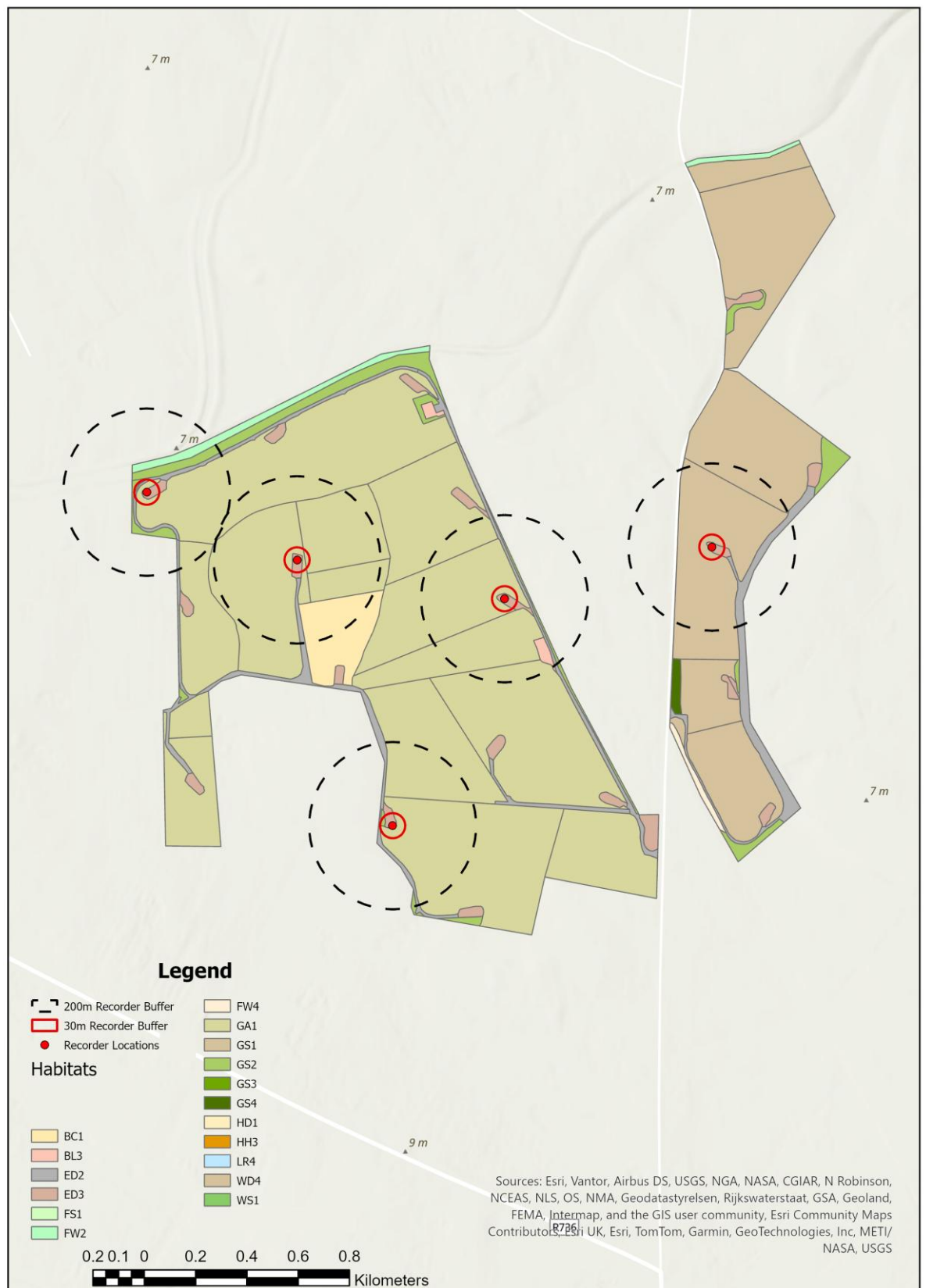


Figure D.2. Acoustic recorder locations on Richfield windfarm. Recorder locations are shown as points, the 30 m spatial buffer is a red circle, and the dashed circle is the 200 m spatial scale.

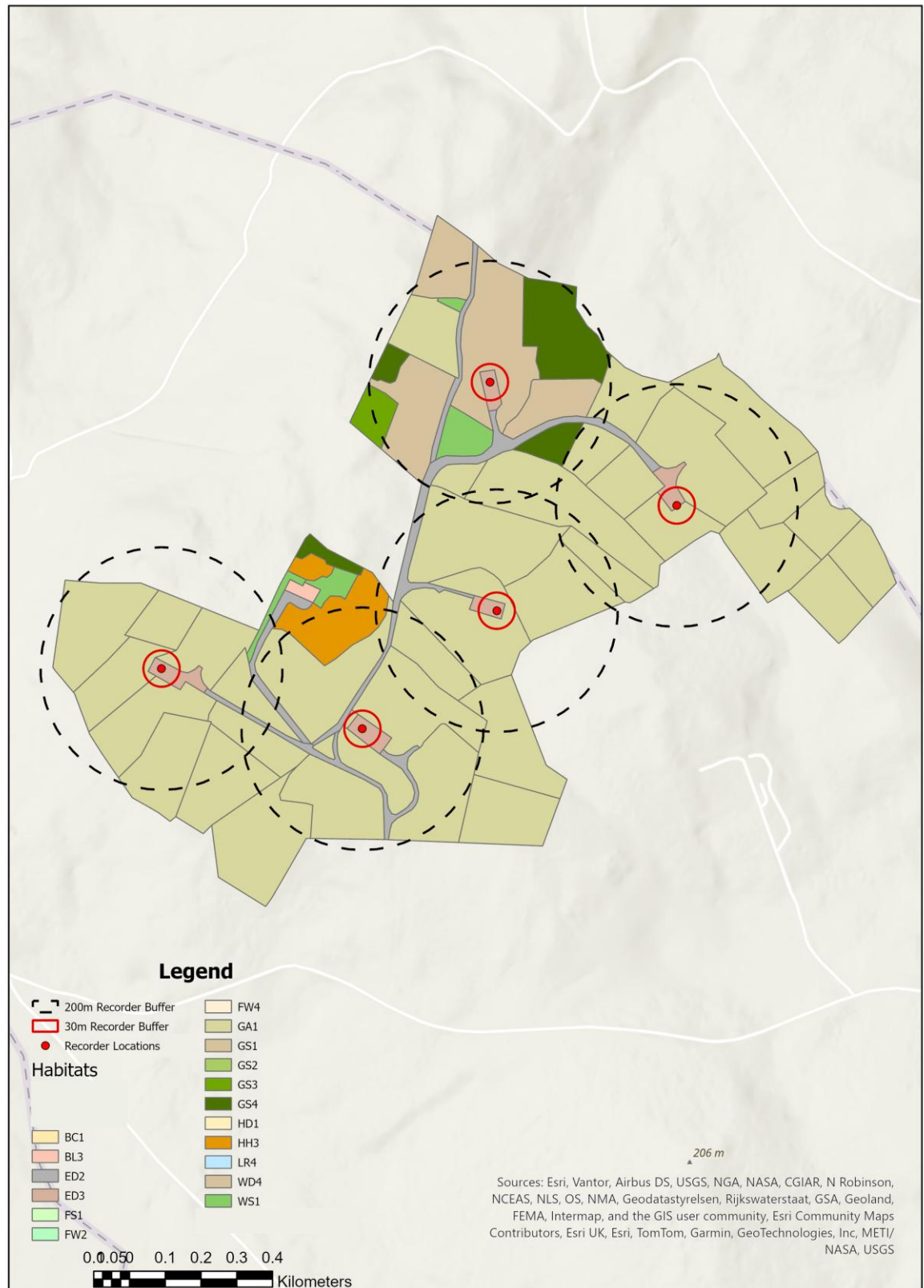


Figure D.3. Acoustic locations on Teevurcher windfarm. Recorder locations are shown as points, the 30 m spatial buffer is a red circle, and the dashed buffer is the 200 m spatial scale.

D2. GLMM Results

Table D.1. Effect of sampling scales (spatial and temporal) on bat activity rates. Significant predictors with a positive effect on bat activity are depicted in blue ($p < 0.05$), and significant predictors with a negative effect are shown in red ($p < 0.05$). NS indicates a non-significant predictor ($p > 0.05$).

	30m			200m		
	10-minute	Nightly	Weekly	10-minute	Nightly	Weekly
Linear feature length	NS ($p = 0.61$)	NS ($p = 0.06$)	NS ($p = 0.07$)	NS ($p = 0.44$)	NS ($p = 0.8$)	NS ($p = 0.39$)
Habitat diversity	NS ($p = 0.78$)	NS ($p = 0.33$)	NS ($p = 0.26$)	NS ($p = 0.66$)	NS ($p = 0.72$)	NS ($p = 0.66$)
Temperature	Significant ($p = 0.04$)	Significant ($p < 0.01$)	Significant ($p = 0.02$)	Significant ($p = 0.04$)	Significant ($p < 0.01$)	NS ($p = 0.06$)
Rain	NS ($p = 0.25$)	Significant ($p = <0.001$)	NS ($p = 0.86$)	NS ($p = 0.22$)	Significant ($p < 0.01$)	NS ($p = 0.72$)

Table D.2. Effect of sampling scales (spatial and temporal) on bat foraging. Significant predictors with a positive effect on bat activity are depicted in blue ($p < 0.05$), and significant predictors with a negative effect are shown in red ($p < 0.05$). NS indicates a non-significant predictor ($p > 0.05$).

	30m			200m		
	10-minute	Nightly	Weekly	10-minute	Nightly	Weekly
Linear feature length	NS ($p = 0.32$)	Significant ($p = 0.008$)	Significant ($p < 0.01$)	NS ($p = 0.24$)	NS ($p = 0.91$)	NS ($p = 0.63$)
Simpson's diversity of habitats	NS ($p = 0.99$)	NS ($p = 0.16$)	NS ($p = 0.08$)	NS ($p = 0.33$)	NS ($p = 0.99$)	NS ($p = 0.94$)
Temperature	Significant ($p < 0.001$)	Significant ($p = 0.02$)	Significant ($p < 0.001$)	Significant ($p < 0.001$)	Significant ($p = 0.04$)	Significant ($p < 0.01$)
Rain	Significant ($p = 0.01$)	Significant ($p = 0.03$)	Significant ($p < 0.001$)	Significant ($p = 0.01$)	Significant ($p = 0.03$)	Significant ($p < 0.01$)

Table D.3. Effect of sampling scales (spatial and temporal) on bat species richness. Significant predictors with a positive effect on bat activity are depicted in blue ($p < 0.05$), and significant

predictors with a negative effect are shown in red ($p < 0.05$). NS indicates a non-significant predictor ($p > 0.05$).

	30m			200m		
	10-minute	Nightly	Weekly	10-minute	Nightly	Weekly
Linear feature length	Significant ($p < 0.01$)	Significant ($p < 0.01$)	Significant ($p = <0.01$)	Significant ($p < 0.01$)	NS ($p = 0.13$)	Significant ($p = 0.04$)
Simpson's diversity of habitats	NS ($p = 0.3$)	Significant ($p < 0.01$)	Significant ($p = <0.01$)	NS ($p = 0.33$)	NS ($p = 0.71$)	NS ($p = 0.48$)
Temperature	Significant ($p < 0.01$)	Significant ($p < 0.01$)	Significant ($p < 0.01$)	Significant ($p < 0.001$)	Significant ($p < 0.01$)	Significant ($p < 0.01$)
Rain	Significant ($p = 0.01$)	Significant ($p < 0.01$)	NS ($p = 0.34$)	Significant ($p = 0.01$)	Significant ($p < 0.01$)	NS ($p = 0.34$)