An interdisciplinary approach to achieve consensus for evidence-based management of red deer (*Cervus elaphus*) in the Lake District

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Submitted in accordance with the requirements for the degree of

Doctor of Philosophy

The University of Leeds

School of Biology

October 2025

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The candidate, Thomas William Logan, confirms that the work submitted is their own, except where work which has formed part of jointly authored publications has been included. The contribution of the candidate and the other authors to this work has been listed explicitly below. The candidate confirms that appropriate credit has been given within the thesis where reference has been made to the work of others.

Chapter 3

I was responsible for conceiving and designing the study, including aims and objectives and methods. I collected and analysed the data, wrote the manuscript and was responsible for subsequent editing. CRH assisted with workshop facilitation. Supervisors (AIW, RJP, CRH) scrutinised draft manuscripts, providing editorial and conceptual comments to improve the work. Co-authors (AIW and CRH) gave permissions for this chapter to be submitted for peer review in ‘Biological Conservation’, and subsequent publication.

Acknowledgements

This work was supported by the Leeds-York-Hull Natural Environment Research Council (NERC) Doctoral Training Partnership (DTP) Panorama under grant NE/S007458/1. There was further logistical and financial support from my collaborative partners in the National Trust, United Utilities, and Natural England – a huge thank you to those organisations for all their support. This PhD studentship began at the University of Hull before transferring to the University of Leeds near the end of my first year. Therefore, I offer my thanks to the Energy and Environment Institute at the University of Hull for their early support.

I would like to extend my gratitude to the Lake District National Park Authority for providing an independent venue for the stakeholder participatory workshop in Chapter 3, and the North York Moors National Park Authority for allowing me access to their land for surveys in Chapter 5.

To my outstanding supervisory team (or my roadies, as I once called the four of you whilst presenting at a conference in Alaska, after a moment where I may or may not have publicly compared myself to Mick Jagger …).

Dr Alastair Ward. I have no idea how to put into words how grateful I am for the near decade that you have invested in me – BSc, MSc, PhD … you've seen me through it all. You are a fantastic mentor driven by passion and enthusiasm who has both inspired and helped me beyond measure since we first met. One of the greatest decisions I ever made was asking you to supervise my masters in 2017. For me, it began the greatest working relationship, and education, that I could ever have hoped for. I will be forever grateful to you for your guidance, patience, support, enthusiasm, good humour, and friendship. JFDI.

Dr Charlotte Hopkins. My social science whizz. I have often joked since starting my PhD that all deer management researchers should acquire highly qualified marine biologists to ensure success. Joking aside, in this instance, every word of it was true. I simply cannot comprehend how the social science elements of this project could have worked without your guidance, passion, wisdom, and encouragement; all of which was shared with a smile, kindness, lots of laughter, and lots of coffee (I must get hold of that banana bread recipe from you …).

Dr Steve Sait. I'd like to apologise for the distinct lack of entomological stimulation for you, although I am sure my constant moaning about ticks will have scratched that particular itch. I am extremely grateful for you joining the project, facilitating my transfer to the University of Leeds from Hull, and for the efforts, advice and inputs you have provided for me. And of course, for the occasional chat about the cricket when I'd knock on your office door.

Professor Rory Putman. It has been a distinct pleasure to work with **the** Rory Putman. One of my many reasons for applying to this project was the opportunity to work with you. Your knowledge, philosophy, guidance, experience, and ruthless challenging of me have pushed me in the best way that I could ever have asked for. I am incredibly grateful for the time you have given me in your retirement, and I feel privileged to be able to call you a colleague and friend. I look forward to continuing our captivating phone calls in the future.

I owe an unbelievably enormous debt of gratitude to Thomas Harrington, Alistair Walton, Robin Gillespie, Vicky Bowman, Nigel Pilling, Liam Williams and Alastair Boston for their support in this project. Be it with fieldwork, workshops, stakeholder engagements, advice, confidence boosts, logistical support, and laughter. You have all been incredible.

Further thanks to Ben Harrower from BH Wildlife Consultancy for being the drone pilot and operator for Chapter 4 of this project. Thank you for the stimulating discussions, and your professionalism and hard work on surveys.

My gratitude to friends made, and for their support, during our shared PhD journeys: Dr Amy Shipley, Dr Nicky Kerr, Dr Beth Mroz, Zoe Hudson, Hannah Lacy, Dr Ed Gilbert, Felipe Gallardo, Dr Amber Wagstaffe, James White … just a few of many.

Dr Amy Gresham, you have been an absolute legend. With your kind support, amazing friendship, laughter, walks in the Lakes, countless deer discussions, new opportunities, and many rounds of beer and Mario Kart. Thank you!

To my family and loved ones, including those no longer with us. Your support, love, and encouragement, means more to me than I could ever describe.

If there are any whom I have missed from this list then I do apologise, there were so many to thank, but know you all have my gratitude for life. Thank you.

In loving memory of James Logan, Steve Conroy, and Paul Beazley,

Grandad Jim, Uncle Steve, and Grandad Paul.

I could never have achieved this without you,

I hope I made you all proud.

Abstract

Human-wildlife conflict (HWC) is a global conservation problem and a significant driver of species extinction risk. HWC is commonly associated with human-human conflicts that increase the complexity of creating lasting solutions. Resolving both the human-wildlife and human-human elements of these conflicts requires an interdisciplinary approach to evaluate the ecological impacts of the target species and engage key stakeholders. Red deer (*Cervus elaphus*) are managed by humans to mitigate perceived negative impacts to human land-use objectives. The aim of this PhD was to develop a new best practice process by synthesising natural- and social- science methods to achieve collaborative, landscape-scale adaptive deer impact management in the Lake District National Park. The PhD aim was achieved through the engagement of key stakeholders, evaluating the habitat impacts of deer across the landscape, and estimating the population abundance of deer to inform impact management actions.

Key stakeholders across Borrowdale and Thirlmere were engaged in a bottom-up approach, using a modified Delphi process. The Delphi process was employed to evaluate how stakeholders value a cultural rural landscape and native deer populations within it, and to enable equitable discussion to build consensus in co-created adaptive impact management. Crucially, consensus was achieved through a holistic approach in which participants defined the nature of the conflict, articulated shared values and objectives, and collaboratively constructed an adaptive management decision-making framework, demonstrating that value-led facilitation can reconcile divergent land-use priorities in a contested landscape.

To inform adaptive impact management decision-making, the red deer population size was estimated using four different methods: two census and two sampling methods. The Random Encounter Model (REM) was demonstrably the most cost-effective and

precise approach for informing deer management actions. From REM estimates, mean red deer density showed a suggested decline of approximately 5.7% per year over the three-year study period, but statistical support of this trend was marginal.

The impacts of deer were estimated across the landscape using index methods; Deer Activity and Impact (DAI) method for woodlands, and Putman Method for open range habitats. From 2022 to 2024 in Borrowdale and Thirlmere, average impacts in open-range habitats remained moderate to low-moderate and did not change significantly over time, nor did woodland impacts, which remained low-moderate to moderate. Additionally, the DAI and Putman Methods were evaluated against independent assessments of impact to validate their effectiveness in informing impact management. Both the DAI ($r_s(18) = 0.809$, $p < 0.05$) and the Putman Method ($r_s(38) = 0.893$, $p < 0.001$) demonstrated strong positive correlations with independently derived impact estimates. Taken together, the population and impact assessments demonstrate that robust, low-complexity ecological monitoring methods can be aligned with stakeholder-defined objectives to inform adaptive impact management at landscape scale.

This thesis demonstrated that integrating participatory consensus-building with validated, practitioner-friendly ecological monitoring provides a viable and transferable framework for mitigating long-standing HWC. By embedding population and impact monitoring within a socially legitimate decision-making process, adaptive management becomes both technically defensible and politically durable. Facilitated consensus-building was critical to the co-creation of shared objectives, monitoring actions, and deer population control methods, embedding technical decision-making within a socially legitimate governance structure. This approach has direct implications for deer management and conservation policy, indicating that socially legitimate, adaptive management can be operationalised where ecological evidence and stakeholder values are jointly incorporated into decision-making.

Future research should focus on refining key ecological parameters to support practitioner uptake of monitoring methods and evaluating whether the adaptive management framework can be sustained through self-management. Periodic stakeholder re-engagement will be essential to reassess and adapt the framework, ensuring it continues to underpin management policy and contributes to the long-term mitigation of human–red deer conflict in this multi-user landscape.

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Chapter 1

Introduction

1.1. Human-Wildlife Conflict

The International Union for the Conservation of Nature (IUCN) (2020) described human-wildlife conflicts (HWCs) as “struggles that emerge when the presence or behaviour of wildlife poses actual or perceived, direct and recurring threat to human interests or needs, leading to disagreements between groups of people and negative impacts on people and/or wildlife”. HWCs are recognised as a significant driver of increasing extinction risk for many wildlife species (IUCN, 2022) and are listed as a global conservation problem by the UN Convention on Biological Diversity post-2020 (IUCN, 2022). HWCs can occur wherever there are overlaps between human and wildlife populations (Ma et al., 2024). HWCs are reported to have grown in frequency and intensity (Abrahms, 2021; Songhurst et al., 2017) and are becoming more widespread as the human population expands (IUCN, 2022).

HWCs are not limited to any single taxon, and span multiple trophic levels including apex predators, meso-predators, and grazing and browsing species (Nyhus, 2016). HWCs were previously considered to only be the direct conflicts between humans and wildlife, however human-human conflicts that arise in response to HWCs have been clearly identified and are now considered a significant element to managing HWCs (Goyes, 2022; Pimid et al., 2022; Ramos, 2022; Salvatori et al., 2020; 2021). Human-human conflicts in response to HWCs have included significant disagreement between landowners, people and policy makers. Regular law-breaking actions as a result of such disagreements are seen in the form of illegally killing wildlife and violence between people. HWCs have been observed in cases including the revenge killing of predators following livestock or human deaths (e.g. Mersen et al., 2019; Viollaz et al., 2021), and the culling of wildlife implicated in disease transmission to livestock (e.g. Lederman et al., 2021; Naylor et al., 2017). In both examples, the human responses to wildlife-related actions led to human-human conflicts stemming from disagreements over the legitimacy of those actions, and (in extreme cases) escalated to physical violence between individuals (e.g. Mersen et al., 2019; Viollaz et al., 2021).

Currently, the human-human conflicts element of HWCs are studied and managed using a range of social elicitation techniques to engage and facilitate discussion between stakeholders to assist in co-creating conservation and/or conflict mitigation policy (e.g. Amit & Jacobson, 2018; Donfrancesco et al., 2023; Rust, 2017). However, approaches that both evaluate the ecological conditions causing the conflict, and to elicit stakeholders, are needed to be combined into interdisciplinary approaches. This will enable a holistic evaluation that informs and elicits consensus for wildlife management decision-making to enable conflict resolution (e.g. Richardson et al., 2020; White & Ward, 2010).

HWCs between humans and deer are a global concern impacting conservation objectives throughout Europe, North America, Australia, and Asia (Apollonio et al., 2010; Corlatti & Zachos, 2022; Melletti & Focardi, 2025). HWCs between humans and deer are mostly focused on damage to property (Martin et al., 2020; Urbanek et al., 2013), human safety (e.g. road traffic collisions, Bisonette et al., 2008; 2012; Nelli et al., 2018; Putman et al., 2004), disease transmission (Cripps et al., 2019; Waters & Palmer, 2015), agricultural impact (Putman et al., 2011a; Putman & Moore, 1998), habitat regeneration for conservation (Charco et al., 2016; Putman & Moore, 1998; Spake et al., 2020) and competing with other species for finite resources (Corlatti & Zachos, 2024; Ramirez, 2021). This demonstrates the breadth of areas where deer populations may negatively impact anthropogenic activity and land-use objectives, through reducing natural regeneration/habitat restoration, and impacting sources of economic income for people, leading to HWCs. Therefore, deer-human HWCs pose a significant challenge to achieving both landscape development objectives and conservation/habitat restoration targets (Corlatti & Zachos, 2022; Melletti & Focardi, 2025).

1.2. Red deer (*Cervus elaphus*)

The family Cervidae (deer) are a member of the order Artiodactyla and are divided between the sub-families of the Cervinae and the Capreolinae, with a distribution that ranges from the tropics to the Arctic, and across Asia, Europe, and the Americas. Many deer species are recognised as ecosystem engineers (Corlatti & Zachos, 2022), which are defined as a species whose activity has the capacity to create, modify, maintain, or degrade habitats through altering the physical environment and disproportionately influencing resource availability to other species compared to population abundance (Briones, 2024; Sanders & Frago, 2024).

In the United Kingdom (UK), there are currently six species of free-living deer (Table 1.1.): red deer, *Cervus elaphus*; roe deer, *Capreolus capreolus*; fallow deer, *Dama dama*; sika deer, *Cervus nippon*; Chinese water deer, *Hydropotes inermis*; and Reeve's muntjac deer, *Muntiacus reevesi*, (Corlatti & Zachos, 2022; Putman et al., 2011; Putman, 2024). Of the deer species present, only two (red and roe) are native species in Great Britian (roe are not present on Éire). Many non-native populations of red and roe deer are present in Great Britain due to translocations from Europe following population crashes and deer farm escapes (Clutton-Brock et al., 1982; Ward, 2005; Putman, 2024). Among the remaining species, fallow deer are considered naturalised after introductions post 11th century (Sykes, 2004), while sika, Chinese water deer (a species of conservation importance due to a declining population in their extant range), and muntjac, are Alien Invasive Species (AIS) to the UK (Lowe &

Gardiner, 1975; Clutton-Brock et al., 1982; Smart et al., 2004; Ward, 2005; Richardson et al., 2020).

Table 1.1. The six deer species present in the United Kingdom.

Species Name	Feeding Strategy	Preferred Habitat	Native / Naturalised / AIS
Red deer (*Cervus elaphus*)	Mixed-Feeder	Open range, Woodland	Native
Roe deer (*Capreolus capreolus*)	Browser	Woodland	Native
Fallow deer (*Dama dama*)	Browser	Woodland	Naturalised
Sika deer (*Cervus nippon*)	Browser	Woodland	AIS
Reeve's muntjac deer (*Muntiacus reevesi*)	Browser	Woodland (*Understory Specialist*)	AIS
Chinese water deer (*Hydropotes inermis*)	Grazer	Lowland Wetland	AIS

Note: Species information was extracted from Corlatti and Zachos (2022).

Red deer (Figure 1.1) are a native species extant throughout Europe with isolated populations in the Middle East (Figure 1.2) and are the largest terrestrial animal extant in the UK (Hmwe et al., 2006; Mattioli et al., 2022). The species is listed as Least Concern by the IUCN with an increasing population trend across its current range, although red deer previously extirpated across Israel, Albania, Jordan, Lebanon, and Syria, primarily from hunting pressure and land-use changes (Lovari et al., 2018). This is a gregarious species of deer, with average herd sizes varying between 3-7 animals in woodlands and up to 50 in open range habitats (Mattioli et al., 2022). Female red deer (hinds) average 1.07-1.22m in height and 63-120kg in bodyweight (Mattioli et al., 2022). Hinds and their infants (calves), typically form herds kept within large home ranges, the size of which is influenced by available forage (Clutton-Brock et al., 1982; Mattioli et al., 2022). Parturition (calving) occurs from late May to early July each year, where a single calf (twins are rare) is born (Mattioli et al., 2022). Male red deer (stags) average 1.07-1.37m in height and 90-190kg in weight (Mattioli et al., 2022). Stags typically form nomadic bachelor herds outside of the breeding season (rut) which occurs throughout late September to late October. The bachelor herds migrate to find suitable forage to assist in injury recovery and increasing bodyweight post-rut (up to 30% body weight is lost during rut) (Clutton-Brock et al., 1982; Mattioli et al., 2022).

Figure 1.1. A mixed-sex group of a single stag (with antlers) and two hinds (top), a red deer stag (middle), and a hind (bottom) from trail camera images as part of data collection for Chapter 4.

Figure 1.2. Map of the current extant range (gold) of red deer (*Cervus elaphus*) in Europe and western Asia, and isolated alien populations in northern Africa (taken from Lovari et al., 2018).

Red deer are a mixed-feeding ungulate, grazing on open range habitats (including grasslands, heathlands, and woodland ground flora) and browsing in deciduous woodlands (Mattioli et al., 2022). Red deer are commonly linked to environmental impacts through their feeding behaviours and habitat trampling (from large body and herd sizes) across numerous habitat types (Clutton-Brock et al., 1982; Mattioli et al., 2022; Melletti & Focardi, 2025). The impacts from red deer populations have the capacity to facilitate the creation and regeneration of habitats through clearing areas of vegetation from feeding and movement behaviours, and spreading seeds through dung (Mattioli et al., 2022). Through natural feeding and movement behaviours, red deer can positively impact natural ecological processes that can benefit both nature and people (Mattioli et al., 2022). However, red deer are recognised as a key driver of habitat homogenisation in the UK, with high densities linked to reduced woodland regeneration, simplified ground flora, and reduced structural complexity across landscapes (Clarke et al., 1985; Corlatti & Zachos, 2022; Dolman et al., 2010; Holl & Armstrong, 2014). Homogenisation of habitats typically occurs when large bodied species, such as red deer, exceed the ecological carrying capacity (the maximum population size of a species that a given environment can sustain without causing “unacceptable” impacts to habitats and species) of a habitat (Kluger et al., 2016). Further, the distribution of red deer in the UK has expanded in range since 1970 (Ward, 2005), with the most recent estimated population size for red deer in Great Britain being between 376000-420000 animals (Ward, 2007). Considering the ecological impacts measured, and estimated population sizes, red deer have become linked to negative impacts upon biodiversity, and unmanaged red deer are considered to hinder habitat restoration efforts and human focused land-use objectives in the UK (Mattioli et al., 2022; Putman et al., 2011).

In the UK, red deer are considered a charismatic and iconic species of the British uplands (Mattioli et al., 2022), with significant economic value from deer management and stalking enterprises (Nilsen et al., 2007; Pérez-Espona et al., 2013; Putman, 2010; 2024). The species is currently distributed across all nations of the British Isles, with concentrated populations in the Scottish Highlands, the Scottish Lowlands, Northern Ireland, Lake District, North York Moors, Humberhead Levels, Yorkshire Dales, Peak District, East Anglia, and Southwest England (Figure 1.3).

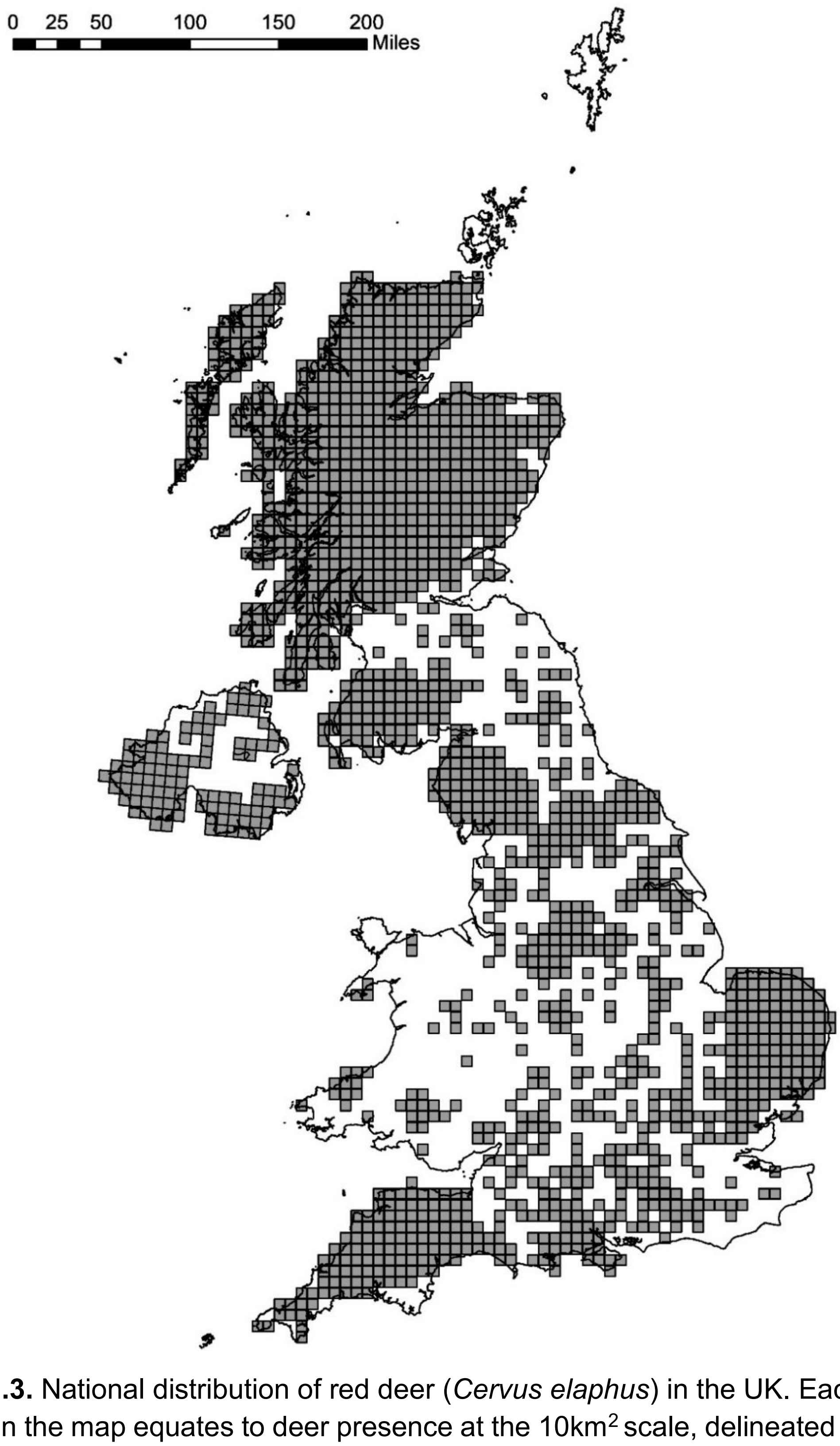

Figure 1.3. National distribution of red deer (*Cervus elaphus*) in the UK. Each greyed square on the map equates to deer presence at the 10km^2 scale, delineated using the British National Grid. The map was created using data shared by the British Deer Society, collected between 1975 - 2022.

Historically in the UK, red deer were predated by grey wolf (*Canis lupus*), with occasional predation from Eurasian lynx (*Lynx lynx*), and brown bear (*Ursus arctos arctos*); predator species which are now extirpated from the UK (Mattioli et al., 2022). Natural mortality of UK deer has since been attributed to predation by meso-predators (juveniles only), disease, age, and food availability (Nilsen et al., 2007). Post-juvenile age-class, there is no natural predation in the UK for red deer. Therefore, there are no predator-prey interactions impacting red deer behaviour (e.g. landscape of fear adjusting habitat use) or recruitment rate (Laundré et al., 2010; Manning et al., 2009; Sand et al., 2021). Thus, wildlife management methods are used in the UK to control population size in order to mitigate HWCs with red deer (Putman 2010; 2024).

1.3. Wildlife Impact Management

Wildlife management is a term used to encompass a wide range of activities that aim to manipulate a population (defined as animals found within a delineated geographic area) of wild animals (Fryxell et al., 2014; Sinclair, 1991). Wildlife management actions can be targeted to suit a range of objectives, including mitigating HWCs (Griffin et al., 2022; Messmer, 2000; 2009; Richardson et al., 2020; Treves et al., 2006). Such activities include adjusting population abundance, influencing habitat use, and harvesting for sustainable yield (examples of manipulative management), and custodial management through monitoring populations for conservation (Fryxell et al., 2014). Selecting wildlife management methods requires technical judgements of the capability of each possible method to meet the established objectives (Fryxell et al., 2014). For example, to mitigate HWCs and illegal poaching of endangered Asian elephants (*Elephas maximus*), wildlife fences have previously been implemented to modify elephant landscape use, reducing agricultural impact from elephants, and thereby addressing the root causes of the HWC (Kochrapa et al., 2024).

In practice, there are five main strategies to manage wildlife, including culling, translocation, fertility control, reintroduction of extirpated species (e.g. predator reintroductions), and exclusion efforts (e.g. fencing or chemical repellents) (Fryxell et al., 2014). Effectively selecting wildlife management strategies is dependent upon numerous factors, including meeting the intended outcomes (Allen et al., 2016; Fryxell et al., 2014; Fuller et al., 2020; Miller & Jones, 2005; Riley et al., 2002), and the social acceptability (Fryxell et al., 2014; Eriksson et al., 2024; Manfredo et al., 2021) of potential management actions, and the potential impacts of those actions on landscape health and conservation objectives (Fuller et al., 2020; Owen et al., 2017; Swan et al., 2017). For example, the culling of ungulate populations to reduce the environmental impacts from target species to meet landscape objectives is an ecologically justifiable management option (Corlatti & Zachos, 2022; Putman, 2024) that has previously been shown to be socially acceptable (Green et al., 2000; Valente et al., 2020a; 2020b). Alternatively, tigers (*Panthera tigris*) are a species of conservation concern that are managed by translocation to preserve individual animals and reduce HWCs from human injury and livestock killing that gains social acceptance (Jhala et al., 2021; Karki et al., 2025; Vasudeva et al., 2021).

Wildlife management actions, such as culling, can be considered socially controversial and difficult to justify ethically (Allen et al., 2023). A key area where this is prominent is in the management of wildlife species classified as endangered by the IUCN that negatively impact anthropogenic activities. The controversies surrounding the culling of tigers, compared to translocations, is a strong example of this (Jhala et al., 2021; Karki et al., 2025; Vasudeva et al., 2021). Further, the killing of animals for sport is considered morally reprehensible by an increasing number of non-hunting societies in European countries (Cohen, 2014; Cohen et al., 2015; Fischer et al., 2013; Von Essen, 2018). In extremities, the killing of animals, in any way and for any reason, has been considered by some as ethically incomprehensible under any circumstances (Allen et al., 2025). Yet, killing animals for food (Byrd et al., 2017; Hoover et al., 2016; Wilkie et al., 2019), to reduce zoonotic disease transmission (Crozier & Schulte-Hostedde, 2014; Lederman et al., 2021), and address cases where species negatively affect habitats and other wildlife (Gordon, 2019; Melton et al., 2021; Robin, 2017), are situations where culling may be considered ethically necessary (Allen et al., 2025). Without understanding the ethical principles, perceptions, and the values of stakeholders to HWCs, it would not be possible to address those deep-rooted values-based problems. Therefore, resolving conflicts requires a clear evaluation of the values of stakeholders before resolutions to conflicts can be facilitated.

Historically, many wildlife management decisions have not accounted for the values and needs of all stakeholders, reducing the chances of consensus in management across landscapes (Kaltenborn et al., 2022). If management decisions are formulated in isolation of other landscape users, the risk of conflict increases (Kaltenborn et al., 2022). For example, the reintroduction of extirpated predators, such as grey wolf (*Canis lupus*), is often proposed as a tool for managing wild deer populations, yet doing so is socially controversial (Sakurai et al., 2024; Wilson & Campera, 2024; Whitehead & Hare, 2025) and evidence suggests it would not be an effective method when compared to the effectiveness of culling to achieve impact objectives (van Beeck Calkoen et al., 2023).

To improve wildlife management actions and outcomes, decision-making frameworks have been developed to facilitate cooperative working between decision-makers with increased consideration for stakeholders, clear objectives by which to measure outcomes, and regular evaluation of data and methods to improve chances of success. Williams (2011a) proposed a framework for adaptive management that encompasses the above needs to increase the likelihood of achieving management objectives, while also reducing human-human conflicts.

1.3.a. Adaptive Impact Management

Adaptive management is a structured process of learning through evaluating management actions and adapting methods and targets, in response to evaluation outcomes, to meet established objectives (Williams, 2011a). The proposed framework from Williams (2011a) differs from previous wildlife management strategies as a more structured, iterative, and learning-oriented approach with an explicit focus on reducing uncertainty to improve decision making, and an emphasis on stakeholder involvement in setting and achieving management objectives. The Williams (2011a) framework was further refined and evaluated into being an interactive cycle of learning information from completed actions to inform future actions in wildlife management (Williams & Brown, 2014; Richardson et al., 2020). Adaptive management has since been recognised as a key factor in improving success rates in wildlife management decision-making (Apollonio et al., 2010; Kingsford et al., 2021; Lynch et al., 2022; Månsson et al., 2023).

Adaptive impact management builds on adaptive management by targeting species impacts to meet predefined landscape objectives, developed and enacted in collaboration with stakeholders (Riley et al., 2003; Williams, 2011a, 2011b). Continuous monitoring and evaluation of data is required to determine progress towards achieving set objectives, with adjustment made by wildlife managers and stakeholders in response to the outcomes from evaluations (Williams & Brown, 2014; 2016). Through evaluations of management actions, evidence-based modifications can be made within the adaptive management process to improve the likelihood of achieving management objectives (Williams, 2011a; 2011b; Williams & Brown, 2016). However, numerous obstacles exist that may impact practitioners implementing adaptive management, for example, institutional inertia, difficulties in stakeholder engagement, being misapplied at different scales, and a resistance to sharing data for evaluations (Williams, 2011a). To overcome these issues, facilitating discussion between stakeholders to identify and reduce these barriers can aid successful implementation of adaptive impact management.

1.4. Consensus building and participation

Within conservation decision-making and HWC mitigation, a substantial body of literature has investigated public values towards nature (Butler, 2016; Raymond, et al., 2016; Byg et al., 2019; Lees et al., 2023; Heindorf et al., 2024; Wardropper, et al., 2024), people's perceptions of conservation problems (e.g. Vatn et al., 2024), levels of agreement with proposed solutions (e.g. Loker et al., 1999; Kilpatrick & LaBonte, 2003; Siemer et al., 2004; Dandy et al., 2011), and the use of the above to inform policy (e.g. Dandy et al., 2009; Hopkins et al., 2018). These approaches have used different methods of engagement (e.g. interviews, questionnaires, workshops), and different analytical tools. However, many previous studies sought to elicit outcomes from individuals rather than facilitating dialogue between stakeholders.

Attempts to create mitigations to HWCs have occurred within the literature (e.g. Byg et al., 2019; Dandy et al., 2011; Kilpatrick & LaBonte, 2003; Lees et al., 2023; Raymond, et al., 2016; Siemer et al., 2004) with successful outcomes concluded against their study aims. Those projects sought to elicit evidence from stakeholders to enable the project teams to create wildlife management decision-making frameworks (e.g. deer management frameworks from Dandy et al., 2009). But there is no clear evidence of continued engagement with those frameworks, which were generic decision-making tools rather than specific to a manageable geographic area. As such, previous research has been limited in creating lasting solutions/mitigations to identified HWCs, often concluding a need for further research to complete the work required to achieve such solutions (e.g. Byg et al., 2019; Dandy et al., 2011; Kilpatrick & LaBonte, 2003; Lees et al., 2023; Raymond, et al., 2016; Siemer et al., 2004). Therefore, the above approaches are rooted in reviewing constituent elements, or narrow research questions (such as identifying perceptions of a conservation problem), rather than adopting a bottom-up, holistic, approach which would enable a single process of defining a perceived problem through to co-creating solutions and/or mitigations.

1.5. Methods of impact control

Wildlife impacts can be broadly defined as the levels and ways in which the activity of wildlife can affect or effect anthropogenic activities (Riley et al., 2002). The control of wildlife impacts can be managed using methods such as exclusion efforts (Hoare, 1992; Jakes et al., 2018; McCollister & Van Manen, 2010), repellents (Bíl et al., 2018; Ward & Williams, 2010), creating wildlife buffer zones/wildlife corridors (Lamichhane et al., 2019; McCollister & Van Manen, 2010), and diversionary supplementary feeding (Murray et al., 2016; Ogden, 2023). Each of the above methods are used to direct wildlife populations away from a resource that is being impacted in a way that is not tolerated. Impact control methods are used in wildlife management to mitigate and reduce the direct effects of a species on a particular area or resource of interest.

For deer management in the UK, impact control methods are divided between using fencing (MacDonald et al., 2022), tree tubing (Marada et al., 2019) and chemical repellents (Fadaei, 2019), all with the aim of reducing impact to sensitive habitats or growing plants. In isolation, the effectiveness of impact control methods on deer in the UK can be limited. For example, deer fences are considered more effective than repellents, but on their own are prone to weaknesses and breaking over time, with deer commonly become trapped within them once gaining entry and thus disproportionately increasing deer impacts within an enclosed area (Iijima et al., 2023). However, deer fences are demonstrably more impactful for reducing deer impacts when combined with population control methods, such as culling (Dolman et al., 2010; Gilbert et al., 2012; Iijima et al., 2023; Walter et al., 2010).

1.6. Methods of population control

Wildlife population control aims to manage the size and growth rate of a population of a species (Stokes, 2011). The common methods used to achieve this include population harvesting (Robinson et al., 2016; Servanty et al., 2011), lethal control through trapping (Littin et al., 2014; Prentice et al., 2019) or shooting (Dandy et al., 2012; Prentice et al., 2019), translocation (Massei et al., 2010; Ueno et al., 2010), or fertility control (Massei, 2023). When considering translocations, there are significant financial costs and welfare concerns with high mortality rate from post-capture myopathy in species that are prone to significant levels of stress (Breed et al., 2019). Therefore, in instances where the species is not of conservation concern, the common alternative method that actively reduces a population size is harvesting and/or culling. Culling, although a socially controversial method (Von Essen & Redmalm, 2023), is the only method that reduces a population size immediately through direct action (Husheer & Robertson, 2005), while the remaining methods can take several years before targeted population reductions are achieved (Massei, 2023; Massei et al., 2010; Ueno et al., 2010).

For deer management in the UK, the only methods for population management used are harvesting and culling by shooting (Putman, 2010; 2024). Translocation is not considered to be an effective method at reducing deer impacts and licences for such actions are rarely granted except in exceptional cases (pers comm. Natural England). Further, under the Deer Act 1991, trapping wild deer (whether to later euthanise or not) requires licensing by Natural England, and is considered a highly stressful practice (Bergvall et al., 2021; Beringer et al., 1996; Huber et al., 2017; Montané et al., 2002). The use of fertility control in the UK is illegal outside of strictly controlled experimental conditions and therefore is not used. Thus the focus in the UK on culling deer for population management assumes fewer deer equates to less impact, despite Moore et al. (2018) concluding that population density only weakly correlates with deer impacts in some habitats. Consequently, UK deer management policy largely equates population reduction with impact mitigation, overlooking the need for targeted actions to directly address ecological impacts.

1.7. Population monitoring and cull targets

Estimating and monitoring wildlife population abundance is primarily divided into two main categories, census counts (an attempt to count all individuals of a population throughout a defined spatial and temporal scale) to express the minimum estimated population size (Kilpatrick et al., 2020), and sampling methods to estimate a maximum-minimum range of population density or abundance (Buckland et al., 2015; Ferretti et al., 2016; Forsyth et al., 2007; Rowcliffe et al., 2008). All methods of estimating population density and abundance assume closed populations with no migration, reducing the reliability of minimum estimated population methods as they are collected across a relatively short time-scale (Kilpatrick et al., 2020). Conversely, sampling methods are likely to capture temporal variation in animal activity, such as migration, due to repeat survey visits (e.g. distance sampling; Buckland et al., 2015), which would likely be reflected in the maximum and minimum range of population abundance.

Ward et al. (2020) demonstrated the impact of uncertainty from estimates of population size and recruitment upon harvest and culling decisions for invasive non-native species (INNS), with only the upper estimates likely to achieve management targets while risking over-harvesting. Following from Ward et al. (2020), combining the maximum estimated population and recruitment rate-based targets may increase successfully achieving objectives for deer adaptive impact management from harvesting and culling deer. In UK deer management, there are few instances where recruitment rate was used to inform management through population modelling (e.g. Ward, 2001), and population abundance is primarily based on minimum estimated population from census methods to inform management actions (Putman, 2010; 2024). Therefore, to meet deer management objectives, cull targets should be informed by precise population estimates and average recruitment rates across targeted landscapes of interest.

1.8. Environmental impact monitoring and indexing methods

Data evaluating the environmental impacts of wildlife populations are a requirement for informing impact management decision-making (Gilchrist et al., 2005; Riley et al., 2002; 2003). For professional ecologists, methods are formulated to evaluate a percentage value of impact from target species upon given habitats (e.g. Cozzi et al., 2019; Melville et al., 1983). The benefits of these methods arise from the statistical evaluation of sampled data to estimate the range of impact across the habitats being monitored. Undertaking such methods is typically resource intensive, requires strict systematic sampling, and requires dedicated time to collect and analyse data (Cozzi et al., 2019). These methods are, therefore, unlikely to be selected by wildlife management practitioners, who are not professional ecologists, due to the complexity and time-consuming nature of the data collection protocols (Maffey et al., 2016; Månsson et al., 2023; Palmer et al., 2021).

Numerous index methods, which summarise impact in broad categories of low-moderate-high, are commonly used to estimate wildlife impacts. With deer impact management in the UK, numerous methods are used throughout the country to estimate deer impacts. These index methods have a range of complexity in data collection protocols, including permanent annual return plots (e.g. Wild Deer Best Practice method), approximately consistent transects (e.g. Deer Activity and Impact method), and randomly selected stations within permanently stratified areas (e.g. Putman Method). The methods are all analysed by descriptive statistics and are capable of capturing broad-levels of wildlife impact, but do not capture fine-scale temporal changes. Further, the methods are in widespread use throughout the UK yet have not been validated to determine their accuracy. As this data is used to inform deer impact management, the accuracy of these methods is essential.

1.9. Aims and thesis structure

This thesis developed an exemplar for interdisciplinary approaches to mitigating deer management related conflicts in the Lake District National Park. To achieve this, the study assessed the biological condition and ecological impact of the deer population and established an exemplar of best practice in collaborative, landscape-scale adaptive deer impact management through the integration of stakeholder values and shared decision-making. A modified-Delphi process was devised to elicit opinions of stakeholders across the Borrowdale and Thirlmere landscape (including landowners, deer managers, residents, farmers and conservation regulators) to understand their perceptions and values towards a healthy landscape, people in the landscape, and red deer in the landscape (Chapter 2), and to facilitate discussion towards consensus on adaptive red deer impact management decision-making (Chapters 3). Additionally, methods to inform adaptive impact management decisions for deer population size (Chapter 4) and deer impacts (Chapter 5) were evaluated to support evidence-based, collaborative decision-making.

The thesis sought to understand to what extent can participatory, evidence-based adaptive management provide a workable framework for managing red deer impacts and conflict in the Lake District National Park.

Chapter 2. Values and perceptions of red deer in Borrowdale and Thirlmere.

Using a modified Delphi process to facilitate discussion between stakeholders in the landscape for this deer population, this chapter sought to elicit the values and perceptions of participants towards landscape health, the people in within it, and the role of deer in the landscape. This chapter used thematic analyses from three-rounds of open questionnaires (focused on the management of red deer) to extract the key values of the participants. Understanding these values identified how participants made the decisions that they did, and provided context that later enabled the creation of objectives for an adaptive management framework for deer impacts at the stakeholder workshop (Chapter 3).

Chapter 3. Consensus based co-creation of an adaptive deer management framework**.**

Using the preceding questionnaires, this chapter focuses on the fourth round of the modified Delphi process. This end-point decision-making workshop brought together the decision-making participants (landowners), the enactors of management actions (deer managers) and conservation regulators. The aim was to facilitate the co-creation of an adaptive management framework for red deer in the landscape, with management objectives informed by values and perceptions from thematically analysed from the earlier questionnaire rounds (Chapter 2). Commitments were made by participants to implement the co-created framework from this chapter to inform deer management policy across the landscape, which future works aim to evaluate.

Chapter 4. Comparative estimates of red deer abundance using four methods.

The population size of red deer in Borrowdale and Thirlmere was measured across three years using multiple methods of both census counts and sampling. This chapter compared on-foot census counts, Unmanned Arial Vehicle (UAV) Drone census counts, line-transect distance sampling, and trail cameras analysed using Random Encounter Model (Rowcliffe et al., 2008). The performance of each method was then compared by the effort to complete, precision of estimates (not available on census), and correspondence of average abundance estimates to Random Encounter Model (the only method performed over all three years).

Chapter 5. Evaluating deer impacts to upland habitats

Current deer impact assessment methods for woodland and open ranges are focussed on quick-assessment index methods. Such methods in the UK have not been previously validated to evaluate whether the methods accurately assess deer impact on an indexed scale, yet these are used for deer management decision-making. This chapter evaluates the Activity and Impact survey method of woodland impact assessments, and the Putman Method of open range impact assessments, with intensively sampled equivalents to evaluate the validity of the methods.

Chapter 6. Discussion.

The thesis concludes with a summary of the main chapters. This chapter integrates insights from the participatory elicitation process and evaluations of estimating deer population size and assessing deer impacts within the English uplands that would not have been possible if the work was performed in isolation. Finally, the chapter identifies future work required to expand upon, and improve, methods of estimating deer population size, evaluating deer impacts, and the need to revisit this case study in the future to evaluate long-term effectiveness of this elicitation process.

1.10. Ethical Statement

All procedures in this thesis were performed in accordance with the ethical standards of the University of Leeds; references BIOSCI 22-002 and BIOSCI 22-004 (see Appendix A). For social science elements of this work, informed consent was given by all participants for their participation and responses to be anonymously presented. All permissions to survey were gained from landowners, and there was no direct contact with animals as part of this research.

Chapter 2

Values and perceptions of red deer in Borrowdale and Thirlmere

At the time of submitting this thesis for examination, Chapter 2 was submitted to journal for peer-review.

Abstract

Rural cultural landscapes are multifunctional systems where ecological health, cultural heritage, and human livelihoods are deeply interconnected. Managing these landscapes requires balancing diverse stakeholder values and addressing potential conflicts over land uses, including conservation priorities.

Here, we explored how stakeholders perceived landscape health and the role of native red deer (*Cervus elaphus*) in the Lake District National Park, a UNESCO World Heritage cultural landscape in the UK. By examining the complex perceptions and values of people towards the landscape, people, and red deer, our research aimed to evaluate how these views may influence landscape management policy to sustain this cultural landscape. As part of a Delphi process, we used three questionnaires to facilitate discussions with participants across Borrowdale and Thirlmere in the Lake District National Park. Using open questions, we thematically analysed responses to evaluate how participants viewed a healthy landscape, the role of people in it, and how red deer influence the landscape.

Participants characterised a healthy landscape as one that balances ecological resilience with sustainable human activity, emphasising habitat restoration, connectivity, and the integration of ecological, economic, and cultural values. Borrowdale and Thirlmere was seen as relatively healthy, with remaining challenges in reconciling conservation goals with sheep farming and tourism, both considered vital economically but potentially harmful ecologically. Red deer were valued as an iconic native species contributing ecological, cultural, and economic benefits, yet their management was deemed necessary to mitigate negative impacts.

Our findings highlight the importance of incorporating community values in landscape management to support the sustained health of rural cultural landscapes. This stands as an example of an effective method to facilitate discussion around contentious issues with stakeholders, and later inform co-created landscape management plans.

2.1. Introduction

Rural landscapes are multifunctional systems that represent examples of varied land use and ecosystem service delivery including food production, forestry, tourism and recreation, cultural heritage, aesthetic value, and nature (Xie et al., 2022; Santoro, 2024). The way that people use landscapes is driven by how they value the landscape (Brunetta & Voghera, 2008; Raymond et al., 2016; Jax et al., 2018; Heindorf et al., 2024). Rural communities are usually considered to have a high level of human-nature connectedness, which is driven by exposure to nature and influences positive associations with conservation actions to maintain nature (Soga & Gaston, 2016; Bashan et al., 2021; Vindevoghel, 2024). In rural areas, people value landscapes, and nature within it, based on a combination of relational values, including aesthetic and cultural value (Jax et al., 2018; Lees et al., 2023; Heindorf et al., 2024; Santoro, 2024; Pan et al., 2025), economic values (Salles, 2011; Raymond et al., 2016; Rea & Munns, 2017; Hanley & Perrings, 2019), and the ecological value (Chan et al., 2016; Santoro, 2024; Pan et al., 2025) from the landscape.

Traditional landscapes have been identified as an essential element in maintaining local cultures and sustaining communities in rural areas (Santoro, 2024). Maintaining traditional rural landscapes, as a central element of both society and economy, would likely promote sustainable landscape use and ecosystem services (Santoro, 2024; Pan et al., 2025). This is related to the actions of land-users (e.g. farmers, foresters) to strengthen the resilience of traditional landscapes, without hindering landscape use (Pan et al., 2025). The UNESCO World Heritage Convention identified the importance of sustaining traditional, cultural, landscapes in 1992 by creating the World Heritage Site category of "cultural landscapes" (defined as the "combined works of nature and of man" (Santoro, 2024)). Traditional cultural landscapes can strengthen the local identity of communities, generate economic benefits through tourism driven by their aesthetic and cultural value, and support natural adaptation to both environmental and anthropogenic disturbances (Pan et al., 2025). Thus, maintaining cultural landscapes is an important element of both nature conservation and community preservation.

Across rural cultural landscapes, perspectives on nature conservation and on how landscapes should be managed vary considerably across communities (Butler, 2016; Raymond et al., 2016; Byg et al., 2017; Lees et al., 2023; Heindorf et al., 2024; Wardropper et al., 2024). Such diversity of views can increase the potential for conflict and reduce opportunities to reach community consensus on landscape management and conservation approaches (Martinez-Juaregui et al., 2023). While many rural stakeholders share a desire to maintain the cultural landscape, their preferred strategies are often shaped by differing, and sometimes conflicting, values (Butler, 2016; Raymond et al., 2016; Byg et al., 2017; Byg et al., 2023; Lees et al., 2023; Heindorf et al., 2024; Wardropper et al., 2024). These values, and significantly varying perspectives, can be influenced by demographic factors (e.g. farmer, homeowner, forester, tourist, land manager). As a result, definitions of a “healthy” landscape, and the acceptable balance between biodiversity, anthropogenic activity, and natural habitat regeneration, are varied (Rapport et al., 1998; Bertollo, 2001; Van der Elst et al., 2018). Therefore, understanding how stakeholders both perceive and value nature and rural landscapes is essential for fostering alignment among land-users and supporting rural landscape conservation.

Healthy landscapes were previously difficult to clearly define, with previous attempts (e.g. National Research Council, 1992) focussed on describing landscapes pre-anthropogenic disturbance. More recently, healthy landscapes have been defined by their ability to support ecological health, and anthropogenic activity (e.g. Rapport et al., 1998; Bertollo, 2001; Van der Elst et al., 2018). The ecological health of a landscape can be judged by the combination of native and non-native biomass present, resilience to natural and anthropogenic disturbance, and supporting of natural processes (Jørgensen et al., 2016; Van der Elst et al., 2018). Within a healthy landscape, people typically value native species more highly than non-natives (Keuffer & Kull 2017; Lewis et al., 2019). In a rural context, therefore, defining the health of a landscape would require judgements on resilience to disturbance, natural processes, and native species persistence.

The Lake District National Park, UK, is designated as a UNESCO world heritage site under the category of a Cultural Landscape (UNESCO, 2025). The landscape is described by UNESCO (2025) as an agro-pastoral land-use system that has been developed by a combination of nature and human activity. The Lake District is a rural mountainous landscape in the Northwest of England with significant cultural heritage, with communities persisting for more than 300 years (UNESCO, 2025). This national park contains numerous connected landscapes with multiple land-uses (e.g. farming, forestry, rewilding, habitat conservation management, housing, tourism) and high levels of native biodiversity. The Lake District is, therefore, a good example of a rural cultural landscape in the UK where conserving the health of the landscape and native biodiversity is essential to maintain cultural heritage.

An example of a native species found in rural landscapes in the UK is the red deer (*Cervus elaphus*), which is the largest extant terrestrial mammal on the British Isles (Mattioli et al., 2022). This species is culturally iconic with a strong aesthetic and economic value to people both within and outside the UK (Scottish Natural Heritage, 2016; Whitefield et al., 2021; Ehrhart et al., 2022). However, the impacts from red deer are not always considered to be beneficial and are a focus for human-wildlife conflicts across rural landscapes (Corgatelli, et al., 2019; Valente, et al., 2020; Ehrhart, et al., 2022). For example, red deer can negatively impact anthropogenic activities (e.g. agriculture and forestry), cause damage to property (e.g. fences, walls), and pose risk to human health through deer-vehicle collisions (Bissonetter et al., 2008; Hothorn et al., 2012; Ahmed et al., 2021; Steiner et al., 2021; Mattioli, et al., 2022; Putman, 2024). Yet, this species can also positively impact people through employment of deer managers and as a food source for people (Mattioli, et al., 2022; Putman, 2024).

In the Lake District, red deer are extant across Borrowdale and Thirlmere, a rural cultural landscape. Across this landscape, >90% of land is owned between National Trust (a conservation charity) and United Utilities (water utilities company for the Northwest of England). There have been conflicts between landowners, including with wider stakeholders (i.e. tenants, farmers, and policymakers), regarding landscape health, conservation actions for protected habitats, and deer management. These conflicts include how the impacts of domestic livestock (a combination of high-density populations of sheep and low-density populations of cattle), and wild populations of native red deer and roe deer (*Capreolus capreolus*), can be managed. In the UK, decision-making for landscape management belongs to landowners, while deer management decision-making rights belong to those who own the shooting rights for the land occupied by deer (usually landowners/occupiers). Yet, wider stakeholders may be impacted by any landscape or deer management decisions made. As such, understanding how people across this landscape value and perceive landscape health, and the presence of people and native red deer, underpins the socio-cultural significance and long-term sustainability of this cultural rural landscape.

In this study we investigated the perceptions and values of people towards nature and deer in the Borrowdale and Thirlmere landscape, including those who work and/or live in the landscape. To evaluate these perceptions, we conducted a series of iterative questionnaires that focussed on the following questions: i) What is a healthy landscape? ii) what is the role of people in a healthy landscape? iii) what is the role of red deer in a healthy landscape?

2.2. Materials and Methodology

We used a Delphi process to elicit the values and perceptions of stakeholders towards the presence and management of a red deer population in a multiple-use modified landscape. The Delphi process is an iterative, usually anonymous, survey with facilitated feedback that allows participants to view the group response and then reflect on and modify their individual response if needed (Mukherjee et al. 2015, 2018). The Delphi process can be used to help resolve conflicts, create a consensus in decision-making and/or highlight a diversity of perceptions and values to inform policy decisions (Mukherjee et al. 2015; Yousuf, 2019). Consequently, the method is readily used by scientists and practitioners in wildlife management and for addressing conservation conflicts (Mukherjee et al. 2015; Rust, 2017; Amit & Jacobson, 2018; Hopkins et al., 2018; Donfrancesco et al., 2023). Here, the use of a Delphi process allowed us to facilitate an evolving discussion that enabled anonymised feedback and reflection to develop and progress this focussed discussion among our participants. This process was not designed to change the views of participants, instead it allowed us to identify the overarching general perceptions and values of participants towards the three questions listed above.

Consistent with Glass et al. (2013) and Hopkins et al. (2018), we developed a stakeholder map (Table 2.1) to identify a range of appropriate organisations and individuals to invite to participate in this study. We used purposive sampling to select participants, with additional participants identified via snowball sampling and communication with partner organisations (Table 2.1). Participants progressed through three questionnaire rounds, which were presented either as electronic documents or hard copy printed document where digital formats were not practical for participants (Figure 2.1). The questionnaire rounds were administered between

September 2022 and February 2023, with participants given six weeks to complete each questionnaire.

Table 2.1. Affiliations and participation of Delphi questionnaire respondents

Participant	Stakeholder affiliation	Questionnaire rounds participated	Identification method
Participant 1	Landowner representative*	1-3	Researcher’s initial network
Participant 2	Landowner representative*	1-3	Researcher’s initial network
Participant 3	Landowner representative* [Retired]	1-3	Referral (*Snowball Sampling*)
Participant 4	Conservation regulator**	1-3	Researcher’s initial network
Participant 5	Conservation regulator**	1-3	Researcher’s initial network
Participant 6	Forestry regulator***	1-3	Referral (*Email Referral*)
Participant 7	Deer manager	1-3	Researcher’s initial network
Participant 8	Deer manager	1-3	Reputation / Referral (*Direct Approach*)
Participant 9	Local resident	1-3	Referral (*Direct Approach*)
Participant 10	Local resident	1-2	Referral (*Direct Approach*)
Participant 11	Tenant farmer	1-2	Referral (*Direct Approach*)
Participant 12	Local resident	1	Referral (*Direct Approach*)
Participant 13	Landowner representative*	1	Referral (*Snowball Sampling*)

Notes: *Landowners for this area are representatives for the National Trust and United Utilities. **Natural England representatives. ***Forestry Commission representative. The reduction of participants over rounds are from survey fatigue rather than selective reductions.

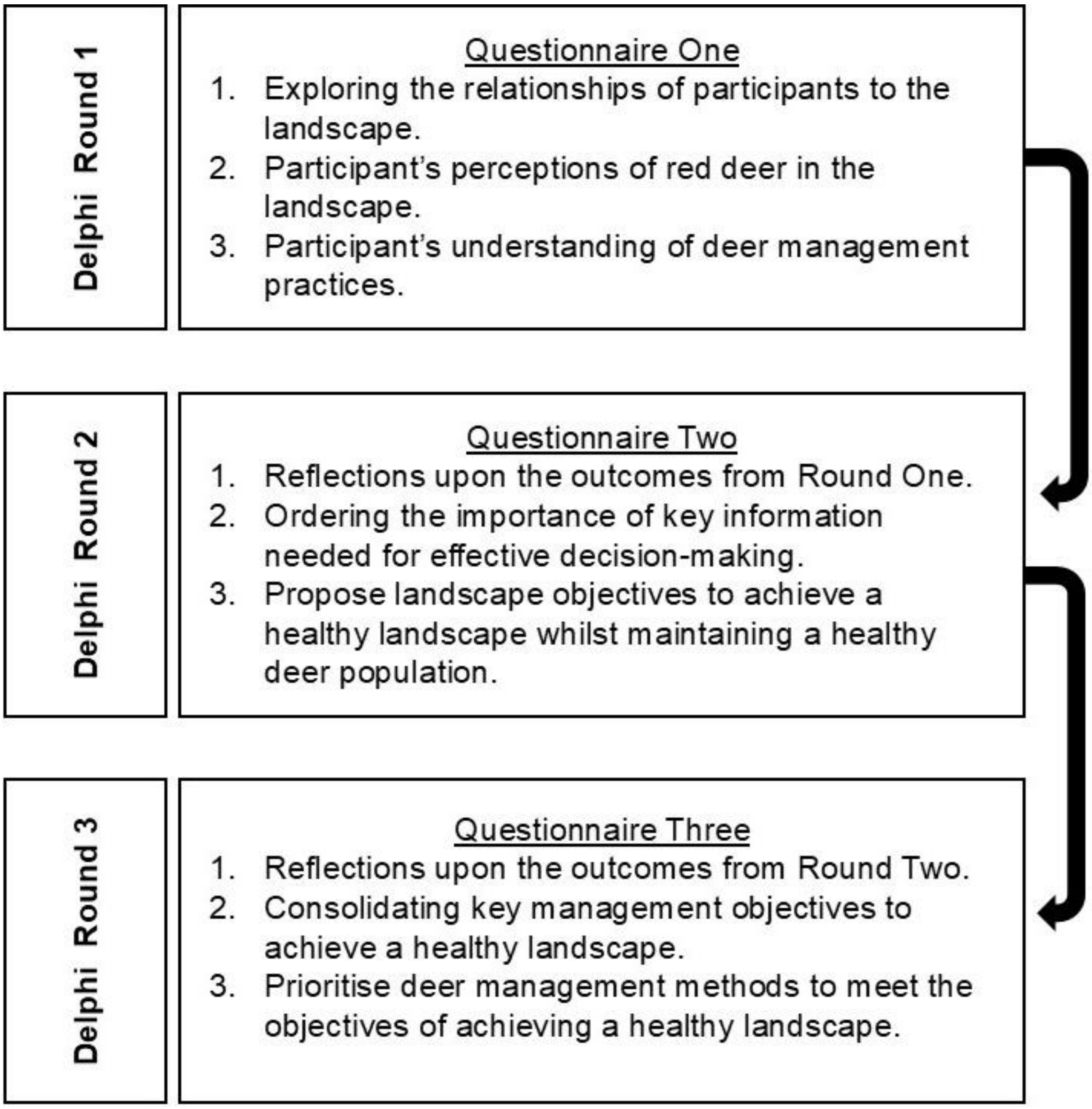

Figure 2.1. Overview of the Delphi questionnaire process to demonstrate how each round was formulated to identify participants' perceptions towards deer management in creating a healthy landscape.

In questionnaire one we aimed to explore participants' perceptions of the landscape, people in the landscape, and the role of red deer (and deer management) in maintaining a healthy landscape in Borrowdale and Thirlmere. We asked participants to respond to a series of open questions (Appendix B), allowing for detailed qualitative expression. We concluded the first questionnaire by asking participants to identify the key information they considered important in making decisions about managing wild red deer.

In questionnaire two, participants were first presented with a summary of responses from the first round, allowing them to reflect and provide additional feedback on key discussions. They were then asked to rank the relative importance of information necessary for effective red deer management to promote a healthy landscape. Following this, participants proposed specific objectives for guiding management actions aimed at achieving a healthy landscape. During analysis, similar objectives were grouped together and summarised to allow participants to comment on them during the feedback element of the final questionnaire.

In questionnaire three, participants were asked to reflect and comment upon a summary of responses to the second questionnaire. In this work, the majority of the data analysis used responses to questions from questionnaires 1-2, and the response to feedback at the beginning of questionnaire 3.

This work focussed on the underlying values and perceptions of participants to questions asked across the three questionnaires. Logan et al. (2025; Chapter 3) later used the findings of this work to inform and focus discussion towards the creation of landscape and deer management objectives as part of an adaptive deer management framework.

2.2.a. Data analysis

Using Microsoft Excel, we conducted a thematic content analysis of the questionnaire data to identify key themes within the dataset (Green & Thorogood, 2014). Thematic content analyses are qualitative analytical tools used to identify, analyse and interpret themes within qualitative data. This analytical tool focuses on the content (what is being said) and the theme (how it is expressed by individuals) to enable social scientists to understand the underlying context and perspectives within the data. For each round, all questionnaires were reviewed and annotated by the lead author to capture emerging themes. Qualitative codes were then identified to capture patterns across participant responses, grouping similar codes under emergent codes. The emergent codes were further refined and categorised under key themes that reflected participants' perceptions for each question. After this initial categorisation, the themes were reviewed by the research team to ensure that each theme was distinct and well-defined. This analytical approach was adapted from the methods recommended by Braun and Clarke (2006) and followed a similar analytical approach to Hopkins et al. (2018).

2.3. Results

2.3.a. Characteristics of participants

A total of 35 people were invited to participate, with 13 participants completing the first questionnaire (37% response rate), 11 continued to complete the second round (85% progression), and nine continued to complete the third round (82% progression). Participant 12 withdrew after the first round due to a perceived lack of knowledge, with three withdrawals (Participants 10, 11, and 13), attributed to survey fatigue. Participants' representation included landowner representatives (31% - n=4), regulatory authorities (23% - n=3), local residents/businesses (23% - n=3), professional deer managers (15% - n=2), and tenant farmers (8% - n=1). Results are summarised in Figure 2.2.

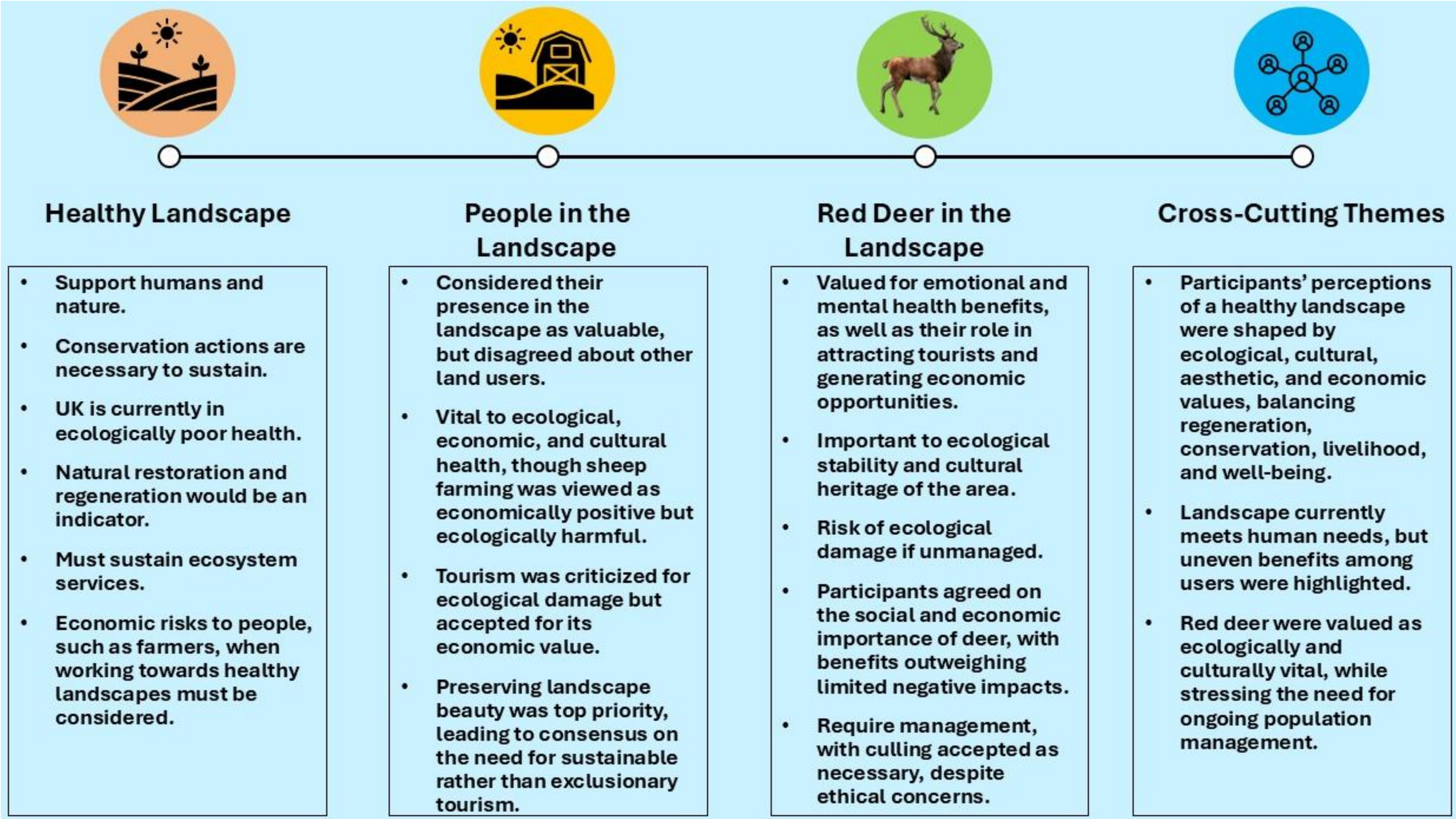

Figure 2.2. Summary of the key perceptions of participants for a healthy landscape, the role of people in the landscape, red deer in the landscape, and the associated cross-cutting themes.

2.3.b. Perceptions of a healthy landscape

Participants agreed that a healthy landscape is one in which humans and nature can be effectively supported and that anthropogenic use of the landscape should not impact ecological stability. A definition for a healthy landscape was proposed as a “resilient landscape for all, delivered through nature recovery and habitat improvement” (**Participant 2**), in questionnaire one. This definition gained support from all participants that chose to comment on it by the final questionnaire; two participants did not comment on this definition. Participants also agreed that there needs to be highly effective conservation and landscape management actions to preserve and enhance the health of landscapes. Most participants stated that the UK is currently in ecologically poor health, and “current conservation [actions are] not working” (**Participant 3**). However, the participants considered the landscape in Borrowdale and Thirlmere to be an example of a landscape where current actions are “mostly effective” (**Participant 9**), although they could be “expanded” upon (**Participant 10**) to “further improve” (**Participant 3**) landscape health.

To ensure a healthy landscape in Borrowdale and Thirlmere, participants suggested that there needed to be evidence of natural “restoration and regeneration” (**Participant 1**) of “resilient habitats and protected sites” (**Participant 4**). To achieve this, an improvement in landscape management was recommended to include “habitats … to be connected” (**Participant 6**) through “planting, rewilding, and regenerating a range of habitats” (**Participant 2**). As such, participants proposed numerous metrics to judge landscape health, including: the landscape “supporting native biodiversity” (**Participant 3**), free of “invasive species” (**Participant 9**), “supporting businesses” (**Participant 6**) such as “farming” (**Participant 11**), and “providing human benefits” (**Participant 13**) reflecting a combination of ecological, cultural and economic values.

Most participants agreed that a healthy landscape must be capable of sustaining ecosystem services without negatively impacting habitats and biodiversity. Ecosystem services, such as food provision, clean air and clean water (**Participant 2, 4 and 13**), were all linked to sustainable landscape use that maximises habitat quality and biodiversity. However, two participants (**Participant 3 and 11**) offered conflicting views that the "constant obsession" with ecosystem services, through the actions of tourism, could damage landscapes and have negative ecological impacts.

Concerns were raised by three participants about the possible economic impacts from only considering the ecological health of the landscape (**Participants 2, 5 and 11**). Anthropogenic activities (e.g. farming, residence, water management) were considered to be reliant upon the landscape being ecologically stable to support their activity. Some participants' solutions to maintain ecological stability included the removal (or restriction) of domestic sheep grazing from the hillside to enable regeneration of natural vegetation. An inability to support sheep farming does not fit the earlier description of a healthy landscape. This demonstrates a persisting disagreement between participants as to whether a healthy landscape can balance biodiversity conservation and some land uses, such as sheep farming.

2.3.c. People in the landscape

All participants self-identified in the first questionnaire as being culturally and economically linked to the landscape through residency, leisure activities, and/or as a source of employment. All participants considered their individual presence as a positive element of the landscape. However, there were inconsistencies between participants as to whether all landscape users could be considered to do so sustainably; for example there were multiple negative perceptions towards the ecological impacts from tourism (**Participant 7, 8 and 11**), and farming (**Participant 3, 5, 7, 8 and 9**). Yet, most participants also agreed that tourism and farming were economically and culturally beneficial to the community across the landscape.

Within the landscape, participants identified their roles individually as being important for ecological health (**Participant 1, 2, 3, 4, 5, 6, 7, 8 and 11**), economic stability (**Participant 1, 2, 11 and 13**), and culturally through professional activity (**Participant 1, 2, 4, 5, 6, 7, 8, 11 and 13**) and providing/owning homes (**Participant 1, 2, 9, 10 and 13**). For example, participants described themselves as “vital for ecological stability” (**Participant 10**) and the “only way to … [protect] biodiversity and the landscape” (**Participant 3**). Yet a disagreement about sheep farming, as to whether current livestock farming and “out of control sheep grazing” (**Participant 8**) in the area was ecologically damaging, persisted throughout the questionnaire rounds, with most participants agreeing that sheep farming was economically beneficial, but ecologically negative in this landscape; encapsulated simply as “it depends” (**Participant 5**).

Most participants perceived negative ecological damage to the landscape from “high levels of tourists” (**Participant 8**), yet participants conceded that the economic benefits are important to the local community. Multiple participants noted growing incidences of unpunished rural crimes (e.g. littering, cutting trees for firewood, illegal fires, poaching wildlife) which were linked to the behaviours of tourists/visitors to the area. With concerns expressed over damage from tourism to the landscape, proposed solutions to mitigate damage were increased education to tourists entering the area, improved infrastructure for tourism activities (e.g. upgraded carparks and repaired pathways), and improved legislation and enforcement for rural crime. These proposals were suggested to “prevent needless damage” (**Participant 11**) to the ecological health of habitats in the landscape. As such, participants demonstrated evidence of perceptions consistent with environmental stewardship and of heritage preservation through approaches to reduce negative ecological impact, whilst supporting economic and cultural benefits from tourism.

It was viewed by the participants that the aesthetic “beauty” of the landscape must be preserved. Three participants (**Participants 8, 10 and 11**) had suggested this may be achieved by removing tourists altogether. However, doing so would be inconsistent with the agreed view from participants, from questionnaire one, that a healthy landscape could support people and viable businesses. As discussions progressed through the questionnaire rounds, it became evident that removing tourists was impossible and would negatively impact the local community. Therefore, participants mostly agreed on the need for tourism and to ensure tourism benefits are maximised without risking the landscape health.

2.3.d. Red deer in a healthy landscape

Red deer were viewed by all participants as essential in the landscape, being described as an “iconic native species” (**Participant 6**) of ecological and “cultural importance” (**Participant 10**). Positive ecological impacts were described from actions from deer behaviour, including “dispersing seeds” (**Participant 3**) as an action “promoting fresh growth and regeneration” (**Participant 4**) in upland and woodland habitats. However, deer were noted as capable of “preventing new growth” (**Participant 6**) and negatively impacting conservation designated habitats through grazing, browsing and trampling. Despite disagreement over current impacts from the red deer, the participants agreed that maximising the ecological benefits of this species is essential for this landscape.

The emotional benefit of the presence of red deer was discussed as a considerably positive attribute. Multiple participants had equated seeing red deer in the area to positive emotional responses and improved mental health, encapsulated by “I love to see them … I wish I had a factual response to back up my emotional wish to see these wonderful animals on my doorstep” (**Participant 9**). The presence of native species in the landscape, including red deer, and the opportunity to see them was linked by some participants as a driver for tourists in the area. As such, the presence of deer, and the joy they bring to people, could directly lead to financial benefits to people in the area (e.g. accommodation, local food provision, outdoor activity retail).

There was a clear agreement from participants about the social and economic importance of red deer in the landscape. People are directly employed across the landscape responsible for the management of red deer (**Participant 1, 2, 3, 6, 7 and 8**) and the regulation of conservation actions that are influenced by red deer (**Participant 4 and 5**). Further, the venison gained from management actions on red deer is sold as food to the local community and through restaurants. There were some negative economic impacts by red deer, although these were minor concerns and did not outweigh the benefits from the deer. Negative economic impacts from red deer included “damage to property walls” (**Participant 11**), “road traffic collisions” (**Participant 13**), and direct resource competition with “[grazing] livestock farming” (**Participant 11**).

To maximise the benefits of red deer in the landscape, all participants agreed that the deer population required managing. There were concerns raised about the methods of deer management. Culling was agreed by all participants to be the only method of effectively managing a population, although individual participants were concerned about ethical considerations. It was proposed that the reintroduction of extinct predators (e.g. Eurasian lynx, *Lynx lynx*) could be attempted as a solution. However, reintroducing predators was eventually rejected by all participants after it raised considerable concern with regards to legality, social acceptance, habitat availability, and risk to pets and livestock.

2.3.e. Cross-Cutting Values

Across three questionnaire rounds, we analysed participants' responses to understand how stakeholders valued and perceived a healthy landscape, the role of people in the landscape and the role of red deer in the landscape. These perceptions were shaped by four key values: ecological, cultural, aesthetic, and economic (Table 2.2). Ecological values emphasised natural regeneration, ecosystem resilience, conservation, and coexistence with people and native species. Cultural and economic views highlighted benefits such as food provision, job creation, tourism, and improved water/air quality. Aesthetic values focused on emotional and mental well-being.

Overall, participants agreed that the landscape currently provides all human needs, and does support some native wildlife and natural regeneration, although this could be improved in the landscape. Participants identified themselves individually to be a key element in this rural landscape, but they did not agree that all landscape users were of equal benefit, suggesting that some users (e.g. tourists) were too ecologically damaging to the landscape. Finally, all participants agreed that red deer must be present in the landscape as a native species, as it has a key ecological role and is of significant benefit to people as a food source, focus of employment, and for emotional well-being. However, participants agreed that the deer population must be continuously managed to prevent and mitigate any negative ecological impact from its presence.

Table 2.2. Summary of participants' responses to the questionnaires relating to a healthy landscape, people in the landscape, and red deer in the landscape in Borrowdale and Thirlmere.

Values	**Healthy Landscapes:**	**People in the Landscape:**	**Red deer in the Landscape:**
Ecological Values The value given to the benefits of biotic and abiotic components for maintaining the environment and biodiversity.	e.g. "Native wildlife are key to contribute to the health of landscapes … linking areas will allow species and habitats to thrive" (**Participant 13**).	e.g. "We work hard to protect and enhance the landscape through planting trees and promoting natural regeneration" (**Participant 10**).	e.g. "A valued part of the native fauna, adding to the biodiversity of the wider landscape" (**Participant 1**). "Deer [are] a vital component of a sustainable and resilient woodland" (**Participant 2**).
Aesthetic Values The value given to the ways in which people experience nature and the natural world both in natural and modified environments. These experiences can be emotional responses, the way in which people are affected, and how people are inspired.	e.g. "[Provides] an inspiring area for people to improve their emotional wellbeing" (**Participant 9**).	e.g. "Sustainable use should not come at the detriment of the visual landscape" (**Participant 1**). "[Too many] tourists spoil the experience of the hilltops" (**Participant 12**).	e.g. "I love to see them … I believe every species is on this earth for a reason and we should seek to protect them … I wish I had a factual response to back up my emotional wish to see these wonderful animals on my doorstep" (**Participant 9**).
Cultural Values The value given to how people interact with, and use, landscapes and wildlife. This could be for housing, recreation, and professional activity.	e.g. "We are lucky to live near this beautiful landscape … [which allows us] to grow much of our own food" (**Participant 10**). 7 participants are employed to maintain the health of this landscape (Table 2.1).	e.g. "Unsustainable use is affecting vast areas of land and affecting the existence of certain species of plants and animals" (**Participant 13**). "Organisations are implementing conservation strategies" (**Participant 7**).	e.g. "An important part of the cultural landscape" (**Participant 5**). "Need to be managed properly [by people] to prevent negative impact" (**Participant 8**).

<u>**Economic Values**</u> The value given to how wildlife and the landscape, and its use, can contribute to, and benefit, human welfare. This can be health benefits, financial benefits, and food provision.	<u>e.g.</u> “Affects water/air quality, soil health … impacting human wellbeing” (**Participant 13**).	<u>e.g.</u> “Tourism plays a major part in the Lake District National Park” (**Participant 13**). “Farming is key for food provision” (**Participant 11**).	<u>e.g.</u> “Red deer provide employment for area and produces a healthy source of meat for the local people” (**Participant 6**).

2.4. Discussion

Using a facilitated Delphi process, we evaluated the values and perceptions of people in a rural landscape in the northwest of England. We identified that all users of this landscape significantly valued the ability of the landscape to simultaneously facilitate the needs of people (e.g. residency, employment, emotional wellbeing) and the ecological health of the landscape. Participants saw themselves, individually, as a significantly important element in maintaining and supporting the ecological, cultural and economic health of the landscape, yet they disagreed that all landscape users were beneficial, with some users seen as having negative ecological impacts (e.g. farmers, tourists) consistent with other works on rural landscape use (e.g. Xie et al., 2022; Santoro, 2024). All participants highly valued the economic, aesthetic and cultural value of red deer in Borrowdale and Thirlmere, although whether the deer in the landscape may be negatively impacting the ecological health of the landscape was discussed and not concluded by the end of our questionnaire process.

By the conclusion of our questionnaire surveys, there was majority agreement between the participants that a healthy landscape should facilitate natural habitat regeneration, support ecological processes and high levels of native biodiversity, and be resilient to disturbance from anthropogenic land-use. The remaining areas of disagreement existed predominantly due to the values that drove the perceptions that led to disagreement (e.g. the economic and cultural value of sheep farming not being compatible with strong ecological health). A continuing source of disagreement, such as farming, is not simple to resolve, and it may not be an area that can be resolved. The majority of participants agreed a definition of a healthy landscape for Borrowdale and Thirlmere would be “[a] resilient landscape for all, delivered through nature recovery and habitat improvement”. The term ‘resilient landscape’ refers to landscapes that support natural process and anthropocentric needs (e.g. Cumming, 2011; Salvati et al., 2013; Eriksson et al., 2018). The agreed definition of ‘resilient landscape’ from our participants is consistent with the definition that emerged from Schippers et al. (2015). The development of this definition was driven by a combination of values that integrate the aesthetic value of a landscape and biodiversity (e.g. Tribot et al., 2018), with the cultural and socioeconomic values of the landscape (e.g. Taylor, 2019; Sanotoro, 2024), and ecological stability (e.g. Schippers et al., 2015; Riechers et al., 2020). Such a definition will lend itself towards actions that meet the intrinsic and cultural needs of people, whilst enhancing the aesthetic and ecological value of the landscape to maintain this rural cultural landscape.

Although participants clearly defined and aspired to a healthy landscape, their perspectives often emphasised its role as a resource for meeting human needs (e.g. employment, food, water). While ecological health was regarded as highly important, discussions frequently framed it in relation to capacity of the landscape to support human well-being and anthropogenic activity. Similar interconnections between anthropocentric and ecological priorities have been observed in other rural values studies (e.g. Mazehan et al., 2013; Ruskule et al., 2013; Csurgó et al., 2021). This dual focus reflects a broader pattern in rural landscapes, where sustaining both human livelihoods and ecological integrity is understood as essential. A clear and consistent theme among participants was that human needs and ecological health should not thrive at the expense of one another, but should instead be realised simultaneously, although this ideal may be difficult to achieve in practice. There is particular importance in this consideration with many remote rural landscapes across Europe face economic underperformance and cultural instability (e.g. Castillo et al., 2024).

Our participant community have varied relationships with the landscape in Borrowdale and Thirlmere, including homes, employment (e.g. landscape managers, farming), leisure activities (e.g. hill walking, cycling, nature spotting), health, and as a resource provider (e.g. food, water, clean air). All of our participants viewed themselves individually as positively impacting the cultural and ecological stability of the landscape, perceiving themselves as essential. Through this, most of our participants have demonstrated attitudes and values towards fostering productive and sustainable relationships with the environment. These attitudes are consistent with the principle of environmental stewardship (see Raymond et al., 2016; Bennett et al., 2018; Bieling et al., 2020) and are commonly identified throughout communities in rural landscapes (e.g. Raymond et al., 2016; Drescher & Warriner, 2022; Mikołajczak et al., 2022; Carmichael et al., 2023). Environmental stewards are commonly associated with environmental restoration and preservation actions as part of their personal and professional activities (Raymond et al., 2016; Drescher & Warriner, 2022; Mikołajczak et al., 2022; Carmichael et al., 2023). Identifying this value within rural landscapes is an important element to nurturing self-driven actions for sustaining cultural landscapes that aim to preserve and enhance the economic, ecological, and cultural value.

There was unanimous agreement that the red deer in Borrowdale and Thirlmere were an essential component of native biodiversity that must exist across the landscape. Participants agreed that the deer positively influence emotional well-being, were a source of employment and food for people, and important to ecological processes in the landscape. A universally positive perspective of red deer is not consistent with the literature, with many sources expressing negative perceptions towards ecological impacts, and socioeconomic impacts (e.g. forestry and agriculture), from deer populations (e.g. Lischka et al., 2008; Johnson & Horrowitz, 2014; Price et al., 2014;

Valente et al., 2020; Hare et al., 2021; Whitefield et al., 2021; Edelblutte et al., 2022; Yu et al., 2024). Our participant community reflected similar concerns with the ecological impacts of deer to the landscape, but with no arable agriculture and minimal forestry in Borrowdale and Thirlmere, socioeconomic impacts were agreed to be positive. Participants' concerns towards the levels of impact from the deer to landscape health, through an overconsumption of resources, led to agreement for the need to manage this deer population to maintain the value of this rural landscape.

The findings from our work have identified that the participant community express a clear desire to maintain the iconic nature of the landscape, including species and habitats. Our participants also clearly expressed the need to maintain the cultural value of this landscape through a combination of anthropogenic actions and natural processes, consistent with the UNESCO definition of a cultural world heritage site. The role of culturally iconic native species, such as red deer, and the health of the landscape (which includes multiple conservation designated habitats) is a significant element of achieving this. The significance of the value of iconic wildlife and environmental structures in rural landscapes is a key area of research to maintain the cultural heritage (Horsley, 2020; Adloo et al., 2023; Santoro, 2024). For example, much of the economic value of rural landscapes, from a tourism perspective, is inextricably linked with the presence, abundance, and preservation of the 'iconic' status of an area, whether related to species, habitats, or geological structures (Paras et al., 2022). Therefore, the desire of our participants to sustainably maintain both the ecological value and human benefits from this landscape is consistent with findings in the wider literature to maintain healthy landscapes, iconic species, and ecosystem services in rural landscapes.

Cultural rural landscapes in Europe are key multifunctional systems involved in a combination of socio-economic benefits and ecological/conservation. However, despite their importance to the cultural heritage of communities, and to environmental management policies, they are in socio-economic and ecological decline across numerous European countries (Castillo et al., 2024). As such, efforts to sustain, manage, and improve socio-economic and ecological stability of rural landscapes is an important conservation action. To maintain cultural landscapes, efforts should be made to improve understanding of how the local community value and perceive the health of their landscape, the iconic species present, and the role of people in that landscape. Actions developed from such understanding would likely increase community engagement and cooperation to increase and promote actions to maintain the rural landscape. In Borrowdale and Thirlmere, we have identified through our Delphi approach that the community highly values the health of the landscape, and the iconic native red deer that reside there.

Our participants valued themselves as an essential element to maintaining and improving the health of the landscape, with significant benefits to be experienced by all through numerous ecosystem services (e.g. food, water, clean air, emotional well-being). As an iconic species, the red deer were considered an essential element to achieving this. However, it was agreed that the deer require managing to maximise their ecological and socio-economic benefit to Borrowdale and Thirlmere. Rural landscapes are demonstrably important across the developed world through their capacity to maintain conservation habitats, food production, tourism, and a cultural hub for communities in countries that are becoming increasingly urbanised. This questionnaire process has demonstrated the ability to evaluate the values and perceptions of the participant community with regards to a shared natural resource across a rural landscape. This questionnaire process demonstrated the ability to evaluate the values and perceptions of a participant community regarding a shared natural resource across a rural landscape. The approach is transferable and may be applied to increase understanding the values and perceptions of stakeholders to landscapes more broadly. As the information may also be inextricably linked to landscape management, the themes draw from such a process may be used in shared decision-making across natural resources. This was demonstrated by our participants who subsequently drew on the themes identified here to co-develop landscape and wildlife management objectives (Chapter 3; Logan et al., 2025).

Chapter 3

Consensus achieving co-creation of an adaptive deer management framework

The chapter has appeared in publication as:

Logan, T.W., Ward, A.I. and Hopkins, C.R. 2025. Stakeholder solutions to human-wildlife conflicts: Co-created adaptive impact management for wild red deer, *Cervus elaphus*, in the English Lake District. *Biological Conservation*. **310**, pp.111380.

Abstract

Human-wildlife conflicts are often symptomatic of underlying human-human conflicts, characterised by opposing viewpoints, unclear or limited communication, and failure to compromise. Participatory approaches towards resolving human-wildlife conflict have previously demonstrated success in creating effective wildlife management plans. In the UK, the red deer (*Cervus elaphus*) is a charismatic native species, yet it is capable of negatively impacting landscape conservation efforts, farming and forestry, and environmental restoration targets.

Here, we employed a facilitated workshop, as the decision-making end point to a modified Delphi process, to foster communication between key interested parties in the Lake District National Park, UK, aiming to produce an adaptive impact management framework for red deer.

Crucially, consensus was achieved for a vision of adaptive landscape management, objectives for the landscape, monitoring actions, and methods of deer population control, and the resulting framework will underpin management policy for this interest group. During this study facilitated consensus-building among interested parties was critical for the co-creation of policy and plans in order to progress towards the resolution of a long-standing conflict with red deer. It stands as an example for interest group engagement with a view to mitigate a human-wildlife conflict in a multi-user landscape.

3.1. Introduction

Human-wildlife conflict can pose critical threats to wildlife populations and is a global risk factor towards species extinction (Göttert & Starik, 2022). The IUCN (2020) described human-wildlife conflict as "struggles that emerge when the presence or behaviour of wildlife poses actual or perceived, direct and recurring threat to human interests or needs, leading to disagreements between groups of people and negative impacts on people and/or wildlife". The frequency of human-wildlife conflict has grown in tandem with increasing urban and sub-urban sprawl into wildlife habitats (König et al., 2020). In Europe, increased human-wildlife conflicts are also linked to the return and increase in species presence (e.g. predator species- Liukkonen et al., 2009; Delibes-Mateos, 2015; Kaiser et al., 2025) and numbers (e.g. ungulates- Apollonio et al., 2010; Delibes-Mateos, 2015; Carpio et al., 2021).

As human-wildlife conflicts increase in frequency and severity, policy interventions have emerged to financially incentivise coexistence (e.g. Naha et al., 2010; Barlow et al., 2013), outlaw activities (e.g. illegal killing) that threaten protected species (e.g. Barlow et al., 2013; Campbell-Palmer et al., 2021), promote the removal of 'problem animals' (e.g. Fukuda et al., 2014; Baker et al., 2024) and minimise negative interactions with people that exacerbate conflicts (e.g. Barlow et al., 2013). However, many of these policy interventions have had limited success, for example Eurasian beaver, *Castor fiber*, in the UK continue to be illegally persecuted near agricultural landscapes despite established co-existence schemes with landowners, and legal protection in Scotland from unlicensed killing (Campbell-Palmer et al., 2021). Unsuccessful attempts at preventing human-wildlife conflict are often a result of failing to address the underlying disagreements between people (Madden & McQuinn, 2014; Bhatia et al., 2020; Woolaston et al., 2021). Persecution of wildlife deemed a problem will therefore likely continue unless the roots of the conflict are directly addressed (Madden & McQuinn, 2014; Richardson et al., 2020; Zimmerman et al., 2020).

Co-created solutions have been proposed as a means of resolving or mitigating such conflicts. These are collaboratively developed by interest groups to design and implement mutually valued outcomes. However, as solutions are very difficult to achieve, and context dependent as to whether eliminating conflict is possible, management tools to mitigate conflict are a more regularly achievable outcome (Redpath et al., 2013; Carter & Linnell, 2016; Nyhus, 2016; Linnell et al., 2020. For instance, human wildlife conflicts between livestock farmers, tigers (*Panthera tigris*), and conservationists have required mitigations that reduce both livestock predation and tiger persecution (e.g. Barlow et al., 2013). Other co-created mitigations have also been used to mitigate and prevent crop damage by Asian elephants, *Elephas maximus*, (Naha et al., 2010). In the above examples, social impacts and conflicts between interested parties were key obstructions to achieving co-created solutions. This is consistent with human-wildlife conflicts being commonly accompanied by human-human conflicts (e.g. Goyes, 2022; Ramos, 2022), thus understanding the social tolerances for wildlife is a key element in conflict resolution/mitigation.

Social tolerances for conflicts with wildlife, and wildlife management methods, are variable across socio-economic categories, such as between developed and developing countries (Dickman, 2010), and rural and urban environments (Kansky et al., 2016). To develop co-created solutions, the variety of social tolerances needs to be clearly communicated (Olsen, 2022) in order to enable equitable discussion (White & Ward, 2010). Social elicitation methods can be used to facilitate those discussions, including evidence-informed discussion of challenging topics, such as lethal and non-lethal wildlife management methods (Smith et al., 2021; van Poorten and Beck, 2021; Cinque et al., 2022; Nelson et al., 2023).

Adaptive management strategies (a process of ongoing planning, management, evaluation and adaptation (Williams, 2011)) have become common practice in wildlife management. A core aspect of the adaptive management framework proposed by Williams (2011), was to include specific considerations for the needs of stakeholders in management planning and action. By those considerations, the social conflicts, and direct wildlife conflicts, can both be addressed more effectively through adaptive management. The flexibility of adaptive strategies enables adjustments for the changing needs of practitioners and for uncertainties in wildlife population parameters (Williams, 2011; Richardson et al., 2020). Adaptive strategies have demonstrated greater success at meeting management objectives compared to static targets, demonstrated in coral conservation (Gelves-Gomez et al., 2024), game harvesting (Nagy-Reis et al., 2021), and invasive species management (Richardson et al., 2020; Davis et al., 2022). Using adaptive management for wild deer populations has been successfully implemented across North America (Conner et al., 2021; Nagy-Reis et al., 2021) and in Europe (e.g. Andersen et al., 2010; Bödeker et al., 2021). In the UK, deer management has been targeted towards mitigating negative impacts to anthropogenic and environmental objectives (Putman, 2010; 2024) across multiple land-ownership boundaries. Yet, government agencies in the UK have reported increased negative impacts from deer, which has resulted in difficulty establishing young trees, impacting forest regeneration, and hindering habitat restoration targets (e.g. Forestry and Land Scotland, 2021; Forestry Commission, 2022). A shift to the adaptive management of deer at the scale at which their populations operate (landscape-scales) might contribute to resolving these conflicts more successfully but requires cooperation among affected interested parties.

In the UK, deer management has become a priority for national policy makers to support environmental restoration policies by reducing negative deer-related ecological and economic impacts (e.g. DEFRA, 2018; NatureScot, 2024). For example, in England WS1 is a voluntary participation, government funded, Environmental Land Management Scheme, administered by the Forestry Commission (2023). The role of WS1 is to financially incentivise landowners to manage deer and monitors deer impacts in woodlands (Forestry Commission 2023). However, there are no legislative powers across the UK (excluding Scotland) to enforce deer management actions to meet public objectives. Further, private landowners may face conflicts between reducing deer populations to mitigate impacts and achieving their own objectives, such as those of private shooting estates (Pepper et al., 2019). Kirkland et al. (2021) previously identified that social conflicts negatively impact deer management outcomes, such that success requires a reduction of social conflicts.

Red deer, *Cervus elaphus*, offer a good example of human-wildlife conflicts in the UK as they range across multiple ownership boundaries throughout landscapes. People are likely to have varying opinions towards red deer populations regarding the presence and impacts of the species. The varying attitudes and values that form these opinions can influence land-use objectives that determine whether impacts are considered positive or negative, affecting property level management actions (Woolaston et al., 2021). Conflicts with red deer can result from grazing, browsing and trampling impacts to areas of conservation concern, arable crops, woodlands and plantation forests, and also from collisions with vehicles (Putman, 2010; 2024; Mattioli et al., 2022).

As a native species, there are efforts to maintain healthy populations of red deer within their extant ranges. To manage wild deer, there are a range of legally permitted methods including culling and exclusion (Deer Act 1991), but these vary in effectiveness, can be context-dependent, and may not be mutually compatible with others (Apollonio et al., 2010). However, under UK laws (Deer Act 1991 in England), the right to take deer belongs to the owners of the shooting rights for the land they are on (normally the landowner). Therefore, in most instances, only landowners have the legal right to take deer, and can make management decisions in isolation with no legal requirement for collaboration with other landowners. Therefore, property level management decisions for red deer can impact neighbouring properties, causing friction between landowners when actions conflict with the values among the local community of interested parties (Ehrhart et al., 2022).

Conflicts surrounding a red deer population exist across neighbouring properties in the Borrowdale and Thirlmere landscape, Lake District National Park, UK. This landscape has an extant native red deer population and is predominantly privately owned by the National Trust and United Utilities. Organisational objectives of the National Trust focus on nature protection, whilst United Utilities prioritises maintaining healthy environments to enable high quality water provisions. Therefore, red deer impact a number of different organisational objectives across the landscape. Most residents, farmers and businesses depend on the landscape, with many in tenancy to the two main landowners. The area also includes seven conservation designated sites (Figure 3.1.) and Borrowdale Rainforest National Nature Reserve. As such, red deer impacts are also of concern to the nature conservation and forestry regulatory authorities, Natural England and the Forestry Commission. With contrasting landscape objectives, conflicts have emerged between key interested parties that may be resolved by co-creating solutions.

Here, we used a modified Delphi process to develop consensus on, and commitment to, adaptive red deer management across the Borrowdale and Thirlmere landscape. We conducted an in-person workshop with a focussed group of key interested parties to consolidate a facilitated discussion, and decision-making on, adaptive deer management. Our objectives were to: i) establish a unified vision for deer management in Borrowdale and Thirlmere; ii) define landscape objectives to support this vision; iii) reach an agreement on the data required to inform management actions; iv) determine the specific management actions to be implemented; and v) develop a framework for adaptive impact management decision-making for the red deer population.

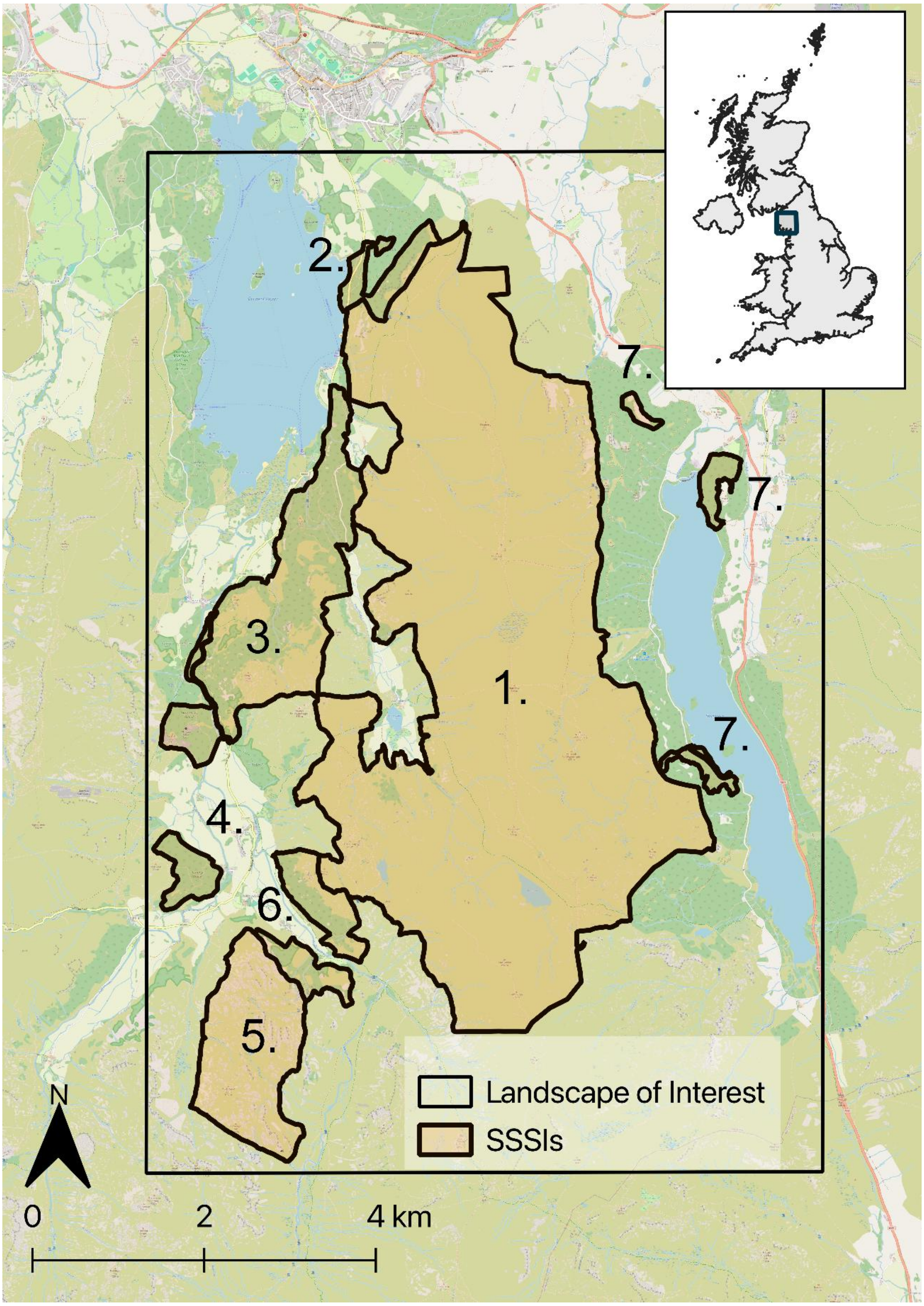

Figure 3.1. Site map of the landscape of interest in Borrowdale and Thirlmere, with an inset map of the UK showing the location of the Lake District National Park via a black squared polygon. The numbered polygons are the seven SSSIs in the landscape: 1) Armboth Fells SSSI; 2) Great Wood SSSI; 3) Lodore-Troutdale Woods SSSI; 4) Johnny Wood SSSI; 5) Rosthwaite Fell SSSI; 6) Stonethwaite Wood SSSI; 7) Thirlmere Woods SSSI.

3.2. Materials and Methods

The Delphi process is a structured and facilitated group discussion technique (usually anonymously) applied to complex topics to create consensus or identify divisions in opinions and decision-making (Linstone and Turoff, 2002; Hsu and Sandford, 2007). The process is also used to incorporate inputs from participants to resolve complex problems, clarify limited and obscure information sources, and resolve conflicts between people (Mukherjee et al., 2015). Application of Delphi is highly adaptable in data collection strategies (e.g. workshops, questionnaires, interviews), application (e.g. level of anonymity), and analyses to suit a wide range of contrasting questions (Yousuf, 2019). Thus, Delphi processes have been increasingly used to resolve problems in wildlife management and conservation (Glass, 2011; Rust, 2017; Amit & Jacobson, 2018; Hopkins et al., 2018; Donfrancesco et al., 2023).

Our Delphi process began in October 2022 and consisted of three emailed questionnaire rounds and a final decision-making workshop in April 2023. Following Glass et al. (2013) and Hopkins et al. (2018), a stakeholder map was created to identify the organisations and interested parties related to deer management in the landscape. The questionnaire rounds invited responses from a wide community of interested parties as part of a wider survey process. 13 participants completed the first questionnaire, 11 completed the second questionnaire, and nine completed the third questionnaire. The withdrawals of two participants per round were attributed to survey fatigue. The questionnaire participants were selected by purposive sampling to represent a wide range of perceptions across the landscape, with additional participants identified by recommendation from partner organisations. Participants comprised: four landowner representatives [1 = questionnaire 1 only; 3 = questionnaires 1-3]; three members of the civil service [questionnaires 1-3]; three local residents [1 = questionnaire 1 only; 1 = questionnaires 1-2; 1 = questionnaires 1-3]; two professional deer managers [questionnaires 1-3]; one tenant farmer [questionnaires 1-2].

These questionnaires sought to facilitate discussion on the perceptions and needs of people for deer management in Borrowdale and Thirlmere. This included: understanding and acceptability of deer management (questionnaire 1); information required and objectives to inform management actions (questionnaire 2); methods to manage deer (questionnaire 3). Participants of the final decision-making workshop were invited if they met one or more of the following criteria: active role in the management of deer in the landscape, decision-making responsibilities for management of deer in the landscape, and regulatory authority regarding nature conservation and woodland management in the landscape.

Six of the seven participants selected for the decision-making workshop (Table 1) were selected from participants of the questionnaire rounds. The seventh was a representative of a participant who was unable to attend the workshop and worked in the same organisation. The workshop participants included representatives of the National Trust and United Utilities (landowners), Natural England and the Forestry Commission (government conservation and forestry regulatory bodies), and two independent contractor deer managers (responsible for planning and enacting culling) for the area. The participants therefore represented a broad scale of professional opinions and experience.

The results of the three questionnaire rounds were presented at the decision-making workshop to inform discussion. Results presented in this paper reflect the final outcomes from the decision-making workshop, informed by the decisions from participants of the earlier questionnaire rounds. Focus group workshops have been used in multiple studies and Delphi processes as a decision-making end point (e.g. MacMillan & Marshall, 2006; Rust, 2017; Hopkins et al., 2018; Segar et al., 2022) and therefore was used in this research to both create decisions and ensure effective cooperation, which will be required once the facilitators are no longer involved.

3.2.a. Data Collection and Analysis

The workshop was structured into three one-hour sessions: i) defining a vision and objectives for the Borrowdale and Thirlmere landscape, and agreeing data needed to measure progress towards the vision; ii) achieving consensus on methods of deer management; iii) producing an adaptive impact management framework for the red deer population. In all the workshop sessions participants used sticky notes to record their observations, responses to other participants' inputs and further thoughts resulting from the discussion. Facilitators offered prompts to individuals for equitable representation, and to the group to keep the discussion focussed and to drive the decision-making process. Sticky notes produced in each session were added to flip charts and were grouped into emergent themes by the facilitators during the workshop (Drinkwater, 2020). Participants were able to comment on the emergent themes, move sticky notes and alter themes in response to the ongoing discussion. Field notes were written by the two facilitators during and after the workshop. Following the workshop, field notes and sticky notes for each session were collated, coded and thematically analysed to identify the emerging themes of discussion from the participants (Braun & Clarke, 2006; Saldaña, 2021).

Table 3.1. Summary of workshop participants and identification method.

Participant	Organisation	Stakeholder type	Identification Method
1	National Trust	Landowner / decision-maker	Researcher's initial network
2	United Utilities	Landowner / decision-maker	Researcher's initial network
3	Natural England	Conservation regulator	Researcher's initial network
4	Natural England	Conservation regulator	Researcher's initial network
5	Forestry Commission	Forestry regulator	Reputation / Referral (Direct Approach)
6	Independent Contractor	Deer Manager	Researcher's initial network
7	Independent Contractor	Deer Manager	Reputation / Referral (Direct Approach)

3.3. Results

3.3.a. Vision and Objectives

The agreed vision for red deer management in the Borrowdale and Thirlmere landscape as a result of the workshop was to: “Create an adaptive impact management framework that is driven by the achievement of established objectives underpinned by high quality data, and delivered in collaboration with key stakeholders, communicated to a wider stakeholder community”. The vision was framed by the agreed outcomes of this workshop, combining the need for adaptive management with the three agreed objectives below. This vision achieved consensus by participants at the end of the workshop.

During the questionnaire process, three recurring themes arose from questions regarding landscape objectives and were agreed by completion of the final questionnaire. The first was the need to ensure that landscape regeneration and biodiversity were key outcomes, without impacting the needs of people. The second was to ensure effective communication between decision-makers and the wider stakeholder community to ensure people in the community were well-informed about deer management. The final theme was to ensure a specific objective that facilitates a healthy population of native red deer that did not hinder landscape regeneration or the needs of people. Based on these principles, the workshop participants established the three objectives below to achieve the vision and meet the themes from the questionnaire process.

Objective 1: “To achieve a healthy, resilient landscape that supports biodiversity, nature recovery, the delivery of ecosystem services and viable businesses”.

Objective 2: "To promote positive and constructive engagement with relevant stakeholders".

Objective 3: "To manage a healthy deer population while ensuring mitigation of negative impacts".

In creating the objectives, a core theme of the participant discussion was ecosystem services and the need to ensure delivery of water quality, food provisioning, healthy wildlife populations and habitats, and public enjoyment of the landscape across Borrowdale and Thirlmere. **Participant 2** stated that a "resilient landscape [was needed] to support habitats and species" and to ensure "resilient catchment for water quality". This was agreed by the other participants, and the importance of "viable businesses delivering for public benefit (including farming)" (**Participant 3**) and "food production [both] agricultural and wild sourced" (**Participant 4**). The importance of "ecosystem recovery [to improve and recover] natural processes" (**Participant 2**) and wider "nature recovery" (**Participant 3**) was also mentioned in relation to the delivery of ecosystem services.

For objective 3, there was consensus amongst the participants that there should be a "healthy red deer population" [within the landscape] (**Participant 3**). It was, however, noted that there may be conflict with farming through the competition of both "deer and sheep [that] needs to be taken into consideration" (**Participant 1**) in management actions. There would likely be a requirement for "new models of farming" (**Participant 7**) to avoid some of these conflicts, and a need for "positive communication with farmers on deer management" (**Participant 5**) to ensure the red deer do not negatively impact people. Participants felt that by managing those relationships and impacts, a well-managed red deer population will be able to exist in the landscape.

To ensure effective use of the agreed framework, all participants agreed that the framework needed to be adaptive and therefore evaluated regularly. By active and regular evaluation, adaptive management actions will readily "respond to [data] changes" indicating a need to "adjust management actions" to meet the desired objectives (Participant 5). However, participants agreed unanimously that any such adjustments to management actions must be agreed by the key landscape interest groups (Table 1) to avoid contradicting Objective 2 and reducing the likelihood of success for Objectives 1 and 3. To further meet Objective 2, deer management decisions would need to be "communicated to the wider [stakeholder] community" (**Participant 1**) to ensure "positive and constructive engagement" (**Participant 2**) throughout the landscape. All decisions on vision and objectives achieved consensus.

3.3.b. Information Needs

Participants of the preceding questionnaires identified, through open questioning, and ranked (scales of 1 = most important – 7 = least important) seven types of information as measurements of the deer population (Table C1) to inform red deer management decision-making. These measurements were presented for discussion at the workshop (Table C1): 1) animal welfare; 2) population size; 3) deer impacts; 4) range and movement; 5) collaborations; 6) population dynamics; 7) traffic collisions. To effectively manage the red deer population across Borrowdale and Thirlmere, participants unanimously agreed that the impacts of deer were key to "informing [the type of] management action" (**Participant 7**) to be undertaken. Specifically, the "impacts from trampling and [herbivory] in woodland and open hill [ecotopes]" **(Participant 3**) would inform management actions (e.g. decreasing a population size).

These deer impacts could be determined through “activity and impact [a quick, industry-standard impact assessment method (Forestry Commission, 2023)]” surveys (**Participant 5**) for consistency with policy makers and other management practitioners.

All participants unanimously agreed that the population size (and to a lesser extent the population dynamics) were needed to “set targets” to achieve the management objectives (**Participant 4**). Participants wanted access to reliable and fast estimates on the population size of deer, with “permanent data that could be [kept and] scrutinised” to assist in creating informed management targets (**Participant 1**). Multiple population size methods were considered by the participants in their discussion, with consensus achieved on annual drone census surveys based on the remit of fast, reliable and permanent records of the deer population. It was also noted that drone census counts would “allow us to understand [sex ratios]” (**Participant 5**), which would satisfy the need to understand population dynamics to inform management. Participants unanimously rejected sampling methods for population estimates, such as modelling data from trail cameras, due to effort, cost, and needing specialist knowledge and training.

Deer welfare (defined by participants as the health of the herd and individual animals) was considered important for meeting objective 3. But it was suggested that welfare did not require specific data collection as “we already get this from culled deer” (i.e. animal weights, parasite load, disease prevalence, and embryos/lactating females) (**Participant 6**). It was unanimously agreed that collating welfare information was needed to, in part, achieve Objective 3, without being considered key to impact management decision-making. The remaining measurements (range and movement, collaborations, and traffic collisions) were unanimously rejected, with minimal discussion, and deemed to be not needed in this framework. All decisions for information needs achieved consensus.

3.3.c. Management Methods

With agreements in place for information needs, the participants then sought to decide which population control methods to use in management (Table C2). The preceding questionnaires identified, through open questioning, and ranked (scales of 1 = most acceptable/effective – 5 = least acceptable/effective) methods for deer management as: 1) lethal control; 2) exclusion and repellence; 3) predator reintroduction; 4) fertility control; 5) translocations.

Lethal control by professional deer managers was the preferred action for all participants as the only method that reduces a population size, and that it should be used to complement strategic exclusion methods. Participants discussed their intention to reduce large-scale exclusion efforts due to their cost and lack of standalone effectiveness. However, "combining lethal control and a combination of tree tubes and fencing can be effective in reducing [browsing] impacts to trees" (**Participant 2**). "These methods should be strategic in their use" (**Participant 1)**, but "only lethal control will work to reduce overall deer impact" (**Participant 5**).

Predator reintroductions were rejected by all participants due to legal and practicality constraints. For example, participants thought that there are not enough interconnected or appropriate habitats for large predators like lynx (*Lynx lynx*), and a high risk of conflict with livestock farmers which would likely lead to persecution of the introduced species. It was, however, noted that it "would be nice" to see such extirpated species "return to Scotland" (**Participant 7**) and "the Lake District"

(**Participant 1**). Translocation of red deer was also discussed and concluded as ineffective by all participants, with **Participant 3** informing the participants that no translocation licences would be issued by Natural England without exceptional circumstances. The final method, which was also unanimously rejected, was the use of fertility control, such as a contraceptive vaccine. This control method was rejected due to legal restrictions, difficulty in administering contraceptives, costs of administering contraceptives, and their usage not fitting the objectives of natural processes in the area. All decisions on methods achieved consensus.

3.3.d. The Adaptive Impact Management Framework

The adaptive impact management framework developed by the workshop participants (Figure 3.2.) established the problem to be addressed, the vision for the landscape in question and agreed objectives to achieve the vision. The framework contains the core requirements of adaptive impact management, with evaluation embedded throughout (see Williams, 2011). Key components of adaptive impact management in this framework include: i) clear objectives; ii) methods of management; iii) population modelling; iv) monitoring data collected; v) evaluation of management; vi) adjustments; vii) stakeholder collaboration and communication. Important considerations for implementing the framework proposed by the participants were to ensure that management decision-making is collaborative and based upon continuously improving information, extracted from monitoring data each month and no longer using static annual targets.

It was further agreed that successful implementation would require receiving a “summary document of the framework” to integrate directly into policy (**Participant 1**).

Further, successful implementation in the community would require “talking to stakeholders that are not here” (**Participant 4**) about “what we have been discussing today” (**Participant 5**), and by openly sharing “our objectives, and how we monitor” the success of management actions (**Participant 2**). Under this framework, red deer population size and dynamics will no longer be calculated from census, instead using annual drone surveys and combined with cull data for population modelling to evaluate previous and future actions. Based on the outcomes of these population models, and monitoring the deer impacts, management actions will be evaluated each month to better meet the objectives set by the participants. With the above agreements, the participants unanimously adopted this adaptive impact management framework for deer impact management policy across the landscape from November 2023.

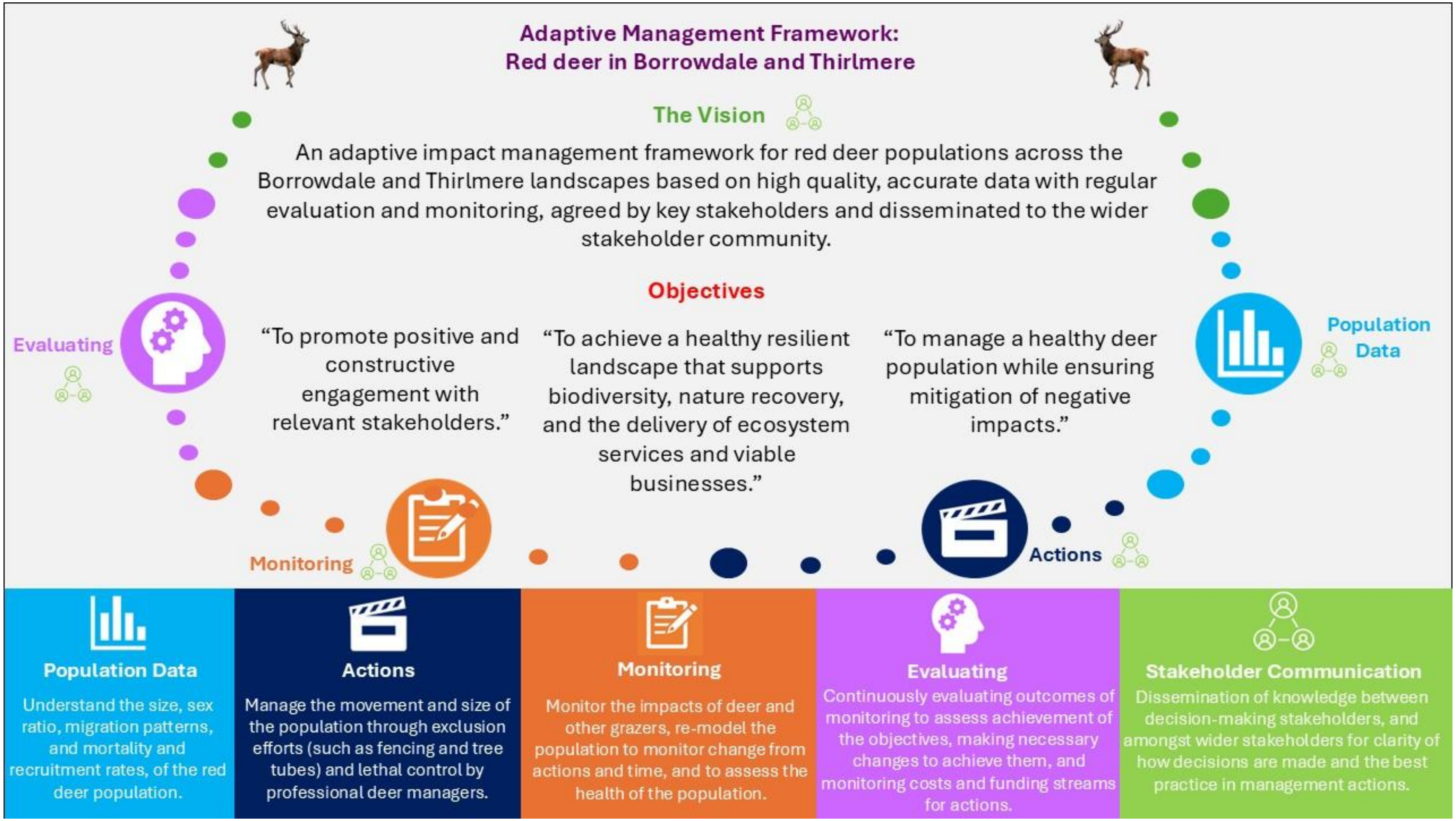

Figure 3.2. Adaptive impact management framework for wild red deer, *Cervus elaphus*, developed by the workshop participant community of key stakeholders across the Borrowdale and Thirlmere landscape.

3.4. Discussion

Landowners in Borrowdale and Thirlmere have been managing red deer in the area for many decades, but existing social conflicts had been impacting relationships with key interest groups across the landscape. The landowners expressed an interest in facilitated engagement with the community to help achieve a deer management solution that could gain local support, initiating our study. Via a facilitated Delphi process, participants achieved consensus for co-created mitigations to conflicts between participants over deer management and landscape-use objectives in Borrowdale and Thirlmere. Without addressing the conflicts between interested parties, effective mitigations for successful deer management would be unlikely to be achieved (Kirkland et al., 2021). Crucially, consensus has been achieved for a unified vision for the landscape, management objectives, management methods, and measurements of success.

Here participants achieved consensus on a co-created vision of adaptive impact management for red deer in Borrowdale and Thirlmere, focusing on habitat recovery actions informed by regularly evaluated data, with decisions made between key interested parties and communicated to the local community. Discussion towards achieving this consensus focussed on combining the needs of people and nature, and effectively communicating throughout decision-making. Considerations of nature and people nature decision-making to improve likelihood of success is consistent with the conclusions from Vatn et al. (2024). Achieving this vision was important for outlining long-term aspirations and providing guidance for developing strategic objectives (e.g. Jones & Kirk, 2018; van de Water et al., 2023) to later guide implementing deer management actions. As vision statements have been demonstrably valuable towards achieving desirable outcomes from conservation efforts (e.g. Merkle et al., 2019), achieving one was considered essential for participants to frame management objectives to inform subsequent efforts (Jones & Kirk, 2018; van de Water et al., 2023), and to reduce institutional obstructions (e.g. Redpath et al., 2013). From the achieved vision, participants gained consensus for three main objectives to measure the performance of management actions, which is a key component of adaptive management (see Williams 2011).

From our workshop, the agreed components within the co-created decision-making framework were mostly reflective of the opinions expressed in the questionnaire process, with few instances of deviating from the priorities of the wider community. The topics of objectives and the priority methods of management from the workshop were directly reflective of the questionnaires; with rejections for translocation, predator reintroduction, and fertility control based on legal restrictions in the UK. Conversely, the agreed information needs were not fully reflective of the questionnaires (e.g. range and movement, and collaborations (defined in Appendix C) were not considered more important than population dynamics by workshop participants). This could impact the community perspective of legitimacy in decision-making (Vatn et al., 2024), which in turn may impact implementation of management actions *in situ* (e.g. Redpath et al., 2013). But discussion from workshop participants ensured decisions were made based on priority data (population size and deer impacts) difficulty (practical and legal restrictions for GPS collaring deer), legal responsibility (traffic collisions), and not considering collaborations to be a measurable information need.

Rejecting range and movement as an information need was surprising from our perspective, as the data would be significantly beneficial to strategic management actions (e.g. strategic culling areas and fencing). Yet, there are practical difficulties and significant expense with GPS collaring deer in the UK. The complexity in obtaining two licences needed to collar deer in the UK, under the Deer Act 1991 and Animals (Scientific Procedures) Act 1986, adds significant difficulty for landowners to consider this. On balance, we would suggest that, if decisions and rationale for them were effectively communicated with the questionnaire participants (consistent with the findings from Brown et al., 2021; Salvatori et al., 2021; Vatn et al., 2024), the findings from this work would likely be acceptable without causing conflict.

Our Delphi process of engagement took a bottom-up approach to participant engagement in human-wildlife conflict. This included allowing the participants to define the problem they perceive, through to facilitating participants in making decisions to achieve a vision and aim that they have defined. This is a novel approach towards conflict mitigation within deer management in the UK, holistically reviewing both the needs of people and nature when facilitating the co-creation of a management decision-making framework to their perceived problem (consistent with Vatn et al., 2024). The need to understand the human element of conflicts in creating effective mitigations is well-documented (e.g., Salvatori et al., 2020; 2021; Pimid et al., 2022), yet few studies have integrated this information into co-created management and conservation action plans. Previous studies have instead commonly pre-determined the problem to be resolved, such as focussing upon whether deer management should occur and the social tolerances for culling (Loker et al., 1999; Kilpatrick & LaBonte, 2003; Siemer et al., 2004; Dandy et al., 2011). We have only found one study in UK deer management combining gathering attitudes and creating a decision-making framework (Dandy et al., 2009), but that framework was designed to inform users whether deer management should occur, rather than how to collaboratively enact management. Such approaches are further reflected in wider literature participatory discussions on human-wildlife conflicts, with focuses upon participant's perceptions (e.g. Loker et al., 1999; Kotulski & König, 2008; Červený et al., 2019; Bavin et al., 2020), or creating frameworks to analyse conflicts (König et al., 2021). We therefore conjecture that our bottom-up approach is an effective advancement to bridge the gap between impacted communities in human-wildlife conflicts and facilitators to co-create strategic wildlife and conservation management strategies.

This Delphi process would also be a useful tool to assist with conservation and government environmental policy actions. For example, recent UK Government environmental and conservation policies (e.g. DEFRA, 2018) necessitate creating clear landscape visions that incorporate both public and private objectives. One of these visions is the creation of the "Great North Forest" in northern England, requiring the establishment of new woodlands through planting up to 50 million trees by 2043 (DEFRA, 2018), which may conflict with alternative land-use objectives. Deer are known to impact both natural woodland regeneration (e.g. Putman & Moore, 1999; Gill & Morgan, 2010; Putman, 2024) and the establishment of new woodlands, which can act as an attractant to deer as a high value resource (Putman, 2010; 2024). Yet the provisions of the Hunting Act 2004, Deer Act 1991, and UK property law, mean that wildlife management decisions are made at property scales. Property scale management decisions risk being less effective than if performed across larger scales (Fattorini et al., 2020), which would require cooperation of multiple interested parties across landscapes. Further, making decisions at only the property scale risks creating and escalating human-human conflicts (Goyes, 2022; Ramos, 2022). To reduce this risk, involving local communities in decision-making helps create and align objectives to achieve management aims across landscapes (Richardson et al., 2020; Pimid et al., 2022). Therefore, achieving environmental and conservation policy actions could be more likely to succeed by using a similar approach to our work.

Despite consensus in agreement by participants to adopt the adaptive management framework as the deer management policy for the landscape, we did not get a clear ‘how’ this agreed policy would be used. The use of facilitators is considered a crucial trusted and impartial figure in conservation planning and mediating conflicts between affected parties (de Vente et al., 2016; White et al., 2023). But once facilitators are no longer involved, and/or if there is a change in trusted personnel, the willingness for participants to follow agreements may reduce (White et al., 2023; Nguyen et al., 2024; Niemiec et al., 2025). Grossman and Patkó (2024) had concluded that the level of disagreement amongst participants had the greatest impact on whether decisions to mitigate human-wildlife conflicts would be adhered to over time. Given the extended opportunities to discuss deer management problems through our Delphi process, we were able to achieve the agreements in the workshop that reached consensus. As such, consensus we achieved may mitigate the impacts of having no continuing facilitators for our participants. However we do not know this, and it should be investigated further to evaluate this lingering question.

The growth in human-wildlife conflicts increases extinction risks for wildlife (Nyhus, 2016), demanding anthropogenic solutions to resolve them (Dickman, 2010). Most human-wildlife conflicts are actually human-human conflicts about wildlife populations (Kansky et al., 2016). Therefore, approaches that address the social conflicts involved in human-wildlife conflicts are necessary for the formulation of co-created solutions and mitigations. Our study highlighted the value of using a modified Delphi process in engaging diverse interested parties to build consensus for a co-developed adaptive management framework for red deer in and upland landscape in England. This consensus building process should be adapted and tested in areas with persisting conflicts to assess whether it will be effective for more contentious case studies of human- wildlife conflicts. Should this prove successful, overcoming social challenges in co-created mitigations to human-wildlife conflicts will be more accessible. Working with key interested parties allows social complexities affecting conservation and wildlife management to be better understood. By balancing the needs of people with landscape health, co-created visions for landscapes can enable social cohesion and allow outcomes that mitigate human-wildlife conflicts.

Chapter 4

Comparative estimates of red deer abundance using four methods

Abstract

Reliable estimates of wildlife abundance are required for effective wildlife management decision-making, yet commonly used methods vary in precision, accuracy, and accessibility. Despite their limitations, wildlife managers typically select census methods in population estimates that, due to methodological choices, cannot quantify uncertainty or a range of population abundance. Using red deer (*Cervus elaphus*) populations in the Lake District as a case study, four population estimate methods were compared to evaluate performance to inform deer management actions.

Red deer population abundance was estimated by two census methods (on-foot counts and UAV drone surveys) and two sampling methods (line-transect distance sampling and trail cameras analysed using Random Encounter Model (REM)). The methods were evaluated and compared based on estimate precision, survey effort, and correspondence between average abundance to identify reliable yet practical techniques for deer population monitoring.

Distance Sampling and REM produced almost identical average estimates, with REM providing a higher precision of estimate range at lower effort compared to Distance Sampling. On-foot census and drone census surveys had a low survey effort, with on foot estimating substantially less deer than REM over two years. Drone census provided inconsistent estimates compared to REM over two years. REM also demonstrated a marginally significant decline in abundance of red deer in Borrowdale and Thirlmere per year from 2022-2024 ($\bar{x}$ = -15.24 (95%CI= -35.12 to 4.63); p = 0.065).

REM produced the most precise and consistent population estimates at relatively low effort. Whilst REM shows potential for practical application, the modelling process needs to be simplified for practitioners, and accurate parameters estimates are essential for model performance. Recent adaptations to REM, such as Random Encounter and Staying Time (REST) may be an alternative to REM from an academic perspective, but the complexity of analysis would not suit practitioner purposes. There were no ideal methods at this stage in the work, but REM, with a simplified analytical process, could be a strong candidate.

4.1. Introduction

Measuring density/abundance of wildlife populations is a continuing challenge in conservation and wildlife management decision-making (Morgan et al., 2024). This includes monitoring changes in population abundance for assessing extinction risk and planning (e.g. Purvis et al., 2000; McCarthy et al., 2014), and populations of overabundant species for wildlife management purposes (Carpio et al., 2021; Morgan et al., 2024). The complexity of methodologies and available technology for estimating population size has been previously noted to have been ‘rapidly’ changing (Lucas et al., 2015). The focus of change was on unbiased survey methods (Morellet et al., 2011) and improving the precision and accuracy of estimates produced, as defined by Barnes (2001) as “a measure of the repeatability of the survey, usually expressed as the standard error or 95% confidence limits” and “whether the estimate is close to the true number of animals” respectively. Improving those elements would thus reduce the risk of statistical uncertainty hindering the development of management objectives (Legg & Nagy, 2006) and subsequent decision-making (Ward et al., 2020; Forsyth et al., 2022). This can be achieved by method choices that produce precise estimates (Kowalewski et al., 2015) and using technology (e.g. thermal imagers) to increase detection probability and reduce uncertainty (Logan et al., 2019).

Population estimates are considered an important element of conservation and wildlife management planning (Legg & Nagy, 2006). Conservation planning will typically use population trends (amongst other factors including extinction risk) to assess whether an intervention is required for a species population (e.g. Bolam et al., 2019). However, in wildlife impact management, other information should also be used (e.g. environmental impacts or species occupancy) to determine whether an intervention is required, with population density/trends then used to inform how an intervention could be implemented (Lyons et al., 2008; Williams et al., 2011; Fuller et al., 2016). In both instances, precise (not defined by the authors) and consistent abundance data on wildlife populations were required for monitoring trends. Yet, non-academic practitioners make up a high proportion of those collecting data, thus requiring 'user-friendly' methods for data collection (e.g. Maffey et al., 2016; Månsson et al., 2023; Palmer et al., 2021). This leads to significant challenges to both scientists and non-academic practitioners in bridging the gap between robust data collection and ensuring the method and analysis is user-friendly for practitioners (advocated by Maffey et al., 2016; Månsson et al., 2023; Palmer et al., 2021). Therefore, to inform effective management decisions, the methods available need to be consistent and balance precise estimates with low complexity to maximise both participation in collecting data, and the quality of data collected.

There are two methodological options to estimate and monitor wildlife population sizes: census counts, and sampling methods. Census counts are a direct observation method attempting to count all individuals of a population throughout a defined spatial and temporal scale (Morellet et al., 2011; Putman et al., 2011b; Chrétien et al., 2015), where results are presented as the minimum population size only. Census counts can be carried out in numerous ways, such as on-foot point or transect counts (Daniels, 2006; Morellet et al., 2011; Putman et al., 2011b; Brunton et al., 2018), and aerial surveys from helicopters (Bender et al., 2003; Daniels, 2006; Ngene et al., 2011; Putman et al., 2011b; Dyal et al., 2022) and drones (Chrétien et al., 2015; Beaver et al., 2020; Duporge et al., 2021). The benefit from aerial counts is the ability to cover large areas/landscapes over a short time reasonably effectively (Ngene et al., 2011; Beaver et al., 2020). However, as census methods are counted over a single short time-period (e.g. one day), and only present a minimum estimated population, there is risk of significantly under-estimating the population from low group detections from animals being either inactive or absent from the area covered on the day (Depraetere et al., 2012; Levy & Lemeshow, 2013), or poor weather conditions impacting wildlife activity. Further, seasonally behaving wildlife populations will have different activity levels across spatial and temporal scales (Depraetere et al., 2012), decreasing the species detection rates (maximised detection rates were advocated for by Field et al., 2007; Petrovan et al., 2011; Logan et al., 2019). In summary, census methods can be significantly impacted by variable detection rates and are restricted to short time-scales, impacting the reliability of data, and do not estimate a range of population size. But, census methods they are quick, have low-complexity, and commonly require low effort from surveyors.

Alternatively, sampling methods are the preferred methods for ecologists when feasible as they combine the data of a subset of wildlife populations and other parameters to extrapolate an estimated range of population size (Levy & Lemeshow, 2013). Sampling can be performed through direct sampling of individual/groups of wildlife (e.g. distance sampling, trail cameras, capture-mark-recapture (Smart et al., 2004; Ward et al., 2004; Lindberg, 2012; Amos et al., 2014; Freeman, 2015; Logan, 2018)), or indirect sampling from signs of wildlife (e.g. faecal standing crop and faecal accumulation rate (Putman, 1984; Mayle et al., 1999; Ward, 2001; Smart et al., 2004; Daniels, 2006)). Whether directly or indirectly sampling populations, sampling methods are conducted over an extended period to maximise detection across varying activity patterns (Levy & Lemeshow, 2013). Such survey efforts can capture temporal variation in wildlife activity (Burton et al., 2015). Increased survey effort across 24-hours of each day surveyed by remote sampling equipment, such as trail cameras, maximises the sampling of species activity to inform population estimates (Burton et al., 2015). Sampling methods can require a high surveyor effort, complex statistical analysis/data modelling, yet, with consistent methodology, sampling can result in high precision and accuracy for population estimates. Consequently, the complexity, effort, and time taken to achieve population estimates could be off-putting to non-academic practitioners (Lorenzini et al., 2022; Maffey et al., 2016; Månsson et al., 2023; Palmer et al., 2021). Yet the estimates from sampling methods are more likely to be considered reliable, with consistent methodology thus leading to well-described population trends to monitor the outcomes of any management actions.

In the United Kingdom (UK), red deer, *Cervus elaphus*, are currently the largest extant terrestrial mammal and considered a native species (Deer Act, 1991; Ward, 2005; Pérez-Espona et al., 2009; Mattioli et al., 2022). Red deer are the focal point of human-wildlife conflicts across many UK landscapes, regularly impacting anthropogenic land-use objectives (Mattioli et al., 2022; Putman 2024). Measuring and monitoring population sizes of red deer is one of the key measurements needed to inform effective management of the species. Across the UK, the most common method used to measure deer population sizes has been census counts conducted by non-academic practitioners, such as deer management groups or game management units (Putman, 2010; Putman et al., 2011b). These census counts have not always been consistently completed, nor have the time-gaps between census counts been consistent (Putman, 2010), reducing reliability of the estimates. Smart et al. (2004) found that population estimate changes below 33% were difficult to detect when estimates were not precise, highlighting the importance of precise estimates to inform management actions.

Sampling methods have been used extensively by ecologists for deer population estimates, including faecal sampling (Putman, 1984; Mayle et al., 1999; Ward, 2001;

Smart et al., 2004), distance sampling (Smart et al., 2004; Ward et al., 2004; Amos et al., 2014), and trail cameras (Freeman, 2015; Logan, 2018). Previously it was demonstrated that repeat measurement methods (including sampling methods) produce more reliable population estimates than single visit methods (Morellet et al., 2011; Putman et al., 2011b). However, comparing the precision of sampling methods to each other, and comparing the estimates available with census counts, has not been completed to our knowledge. Further, factors impacting where, what, and why of methods selected to survey deer is under-studied (Forsyth et al., 2021). Finally, the use of sampling methods for deer population estimates in the UK has not been adopted by industry practitioners, instead continuing to rely upon census data.

We aimed to evaluate whether methods for estimating red deer population sizes could provide reliable data for effective deer management actions. This would be achieved by comparing multiple methods of conducting population census and sampling of a red deer population. The performance of each method would be compared based on effort, precision, and correspondence between estimates. Given the needs of low complexity from practitioners, and high precision for estimates, we would further identify any methods that facilitate this. Therefore, methods that provide reliable and precise population estimates with minimal complexity and low effort would be identified as 'ideal' methods. But, if an 'ideal' method is not found, adequate alternative methods would be discussed based upon a combination of the above factors.

4.2. Materials and Methodology

4.2.a. Study site

This study was carried out in the Borrowdale and Thirlmere sector (Figure 4.1) of the Lake District National Park, UK (Latitude/longitude 54.552677, -3.106075), which ranges between 80-610m above sea level. The area is approximately $40km^2$, of approximately $23km^2$ of open hill ecotopes and $17km^2$ of woodlands. The open hill plant communities comprise primarily of heather moorland and acid grassland. Of the 10 woodlands in the landscape, seven are deciduous Atlantic temperate rainforest, and three are conifer plantation. In Borrowdale and Thirlmere, there are resident populations of red deer and roe deer (*Capreolus capreolus*), sheep and cattle, and numerous species of national conservation importance, such as red squirrel (*Sciurus vulgaris*). The dominant land-uses of the site focuses on conservation management, forestry, livestock farming, anthropogenic recreation (e.g. hillside walking), and reservoir water management.

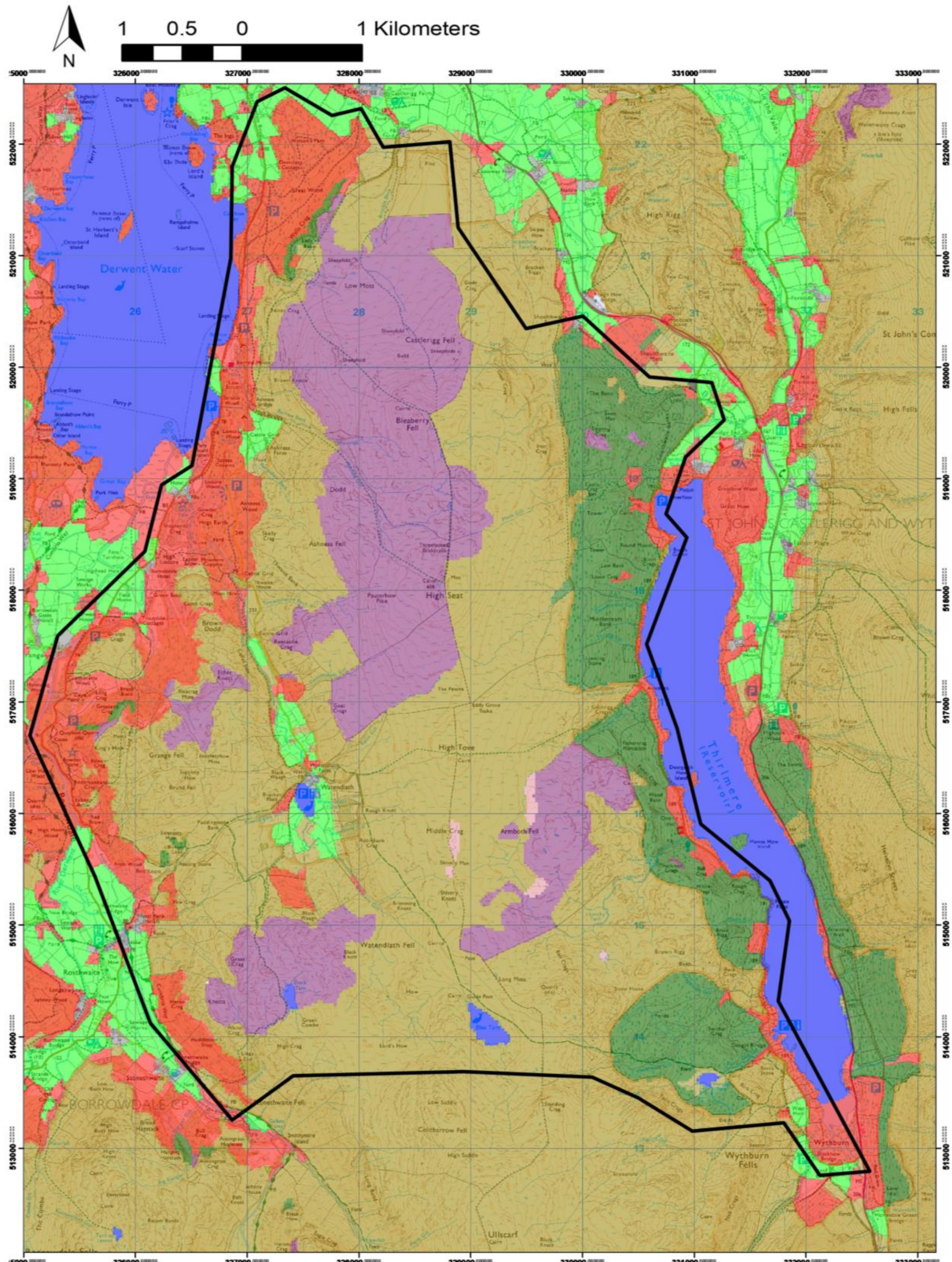

Figure 4.1. The study site in Borrowdale and Thirlmere; the boundary is delineated with a black polygon. Each grid cell on the map is 1km^2, using the British National Grid coordinate system. The Map was developed from open access OS Maps at 1:50000 ratio. The habitat structures were described from aerial surveys, where: purple = wet and dry heather; brown = coarse grassland; light green = improved grassland, dark green = coniferous woodland; red = broadleaf woodland; blue = water.

4.2.b. Estimating red deer population size

Four separate methods were used to estimate the red deer population size in Borrowdale and Thirlmere. These were all direct methods of estimating population size, employing two census methods and two sampling methods (the measurement of a portion of the population and then using statistical methods to estimate the range of population density/abundance).

The performance of each method was compared based on three criteria: the effort to complete data collection (in days), the correspondence between population estimates across each season, and the precision of estimates produced. Precision was analysed using the coefficient of variation, with lower values indicating greater precision compared to higher values. Accuracy was not assessed in this work because the true population size of the red deer was unknown. Although effort was quantified in days, and the cost of equipment for Distance Sampling and trail cameras could be financially quantified, only one method had a clear cost per survey (drone census method), the cost for equipment and time of people for on-foot census counts was not measured and could not be confidently estimated. Therefore, financial cost was not included for the comparison.

4.2.c. On-foot census

In Borrowdale and Thirlmere, on-foot census counts were performed in September 2021 and September 2022 by the landowners and key stakeholders to the red deer population, combining static points and transects depending on the ecotope the individual was surveying. Between 20-30 volunteers were used to count the deer population over a specified 4-hour period starting at sunrise. For each on-foot census count, observations were conducted by a combination of expert deer managers using thermal imagers and binoculars, and amateur volunteer observers with limited prior knowledge. Detection equipment among volunteers varied considerably, including mid-range or low-quality binoculars, and in some cases no detection equipment was used. However, the personnel, experience, knowledge, and equipment used was inconsistent between census counts.

The aim from practitioners and landowners involved in deer management in Borrowdale and Thirlmere is for on-foot census counts to operate as comprehensive counts of all animals in the landscape. However, in practice, these on-foot census counts do not observe every individual, and the annual variation of the proportion of the population detected is unquantifiable between surveys. Double counts of the same animals were removed from the data by comparing the times and locations of observations to the groups counted, completed by the landowners. Final results for on-foot census counts were reported as a total count by the landowners to the study team, rather than raw data, removing the ability to estimate uncertainty.

4.2.d. UAV drone census

In December 2022 and 2023, we used camera-bearing unmanned aerial vehicles (drones) for deer population census counts, piloted by a professional wildlife surveying contractor from BH Wildlife Consultancy (https://www.bhwildlifeconsultancy.com/). The drone was equipped with a thermal camera, a fixed light camera, and a zoom light camera (60x mechanical zoom, 140x digital zoom), with detection distances and positive identifications from up to 3km from the drone.

The operator for the drone census counts surveyed (a structured data-collection exercise) the site using an opportunistic sampling approach. At each drone launch point (Figure 4.2), the operator ascended the drone to a minimum of 45m above the ground and opportunistically sampled the landscape by moving the drone in any direction and using the thermal imager to seek out deer. Upon finding deer, the operator switched between the thermal and light camera to identify the deer species, number of deer, sex, and age class (where possible). These were recorded with permanent photographs to allow the drone operator to process after the surveys and make corrections to the data.

The drone census counts took place across two mornings of 6-hours each (6.30am-12.30pm), with day one on the western side of the study site, and day two on the eastern side. As the surveys were across two days, spatial distribution of the deer may have changed which can impact the population size estimated. This was managed in post survey processing of the images of deer across both days to look for identifiable individuals/groups of deer. For example, if across two days, two groups were detected with one stag, four hinds and three calves, and the stag was identifiable by an asymmetrical antler formation of 7 points on the left antler and 6 points on the right antler, the group would only be counted once. Such post-survey corrections are not infallible but are considered industry standard in UK drone census counts for deer management (Wild Deer Best Practice Guidance for Scotland, 2026).

The results were reported by the drone operator to the landowners as total population abundance, approximate locations of where groups were detected, and an approximate sex ratio. Final results for drone census counts were reported as a total count by the landowners to the study team, rather than raw data, removing the ability to estimate uncertainty.

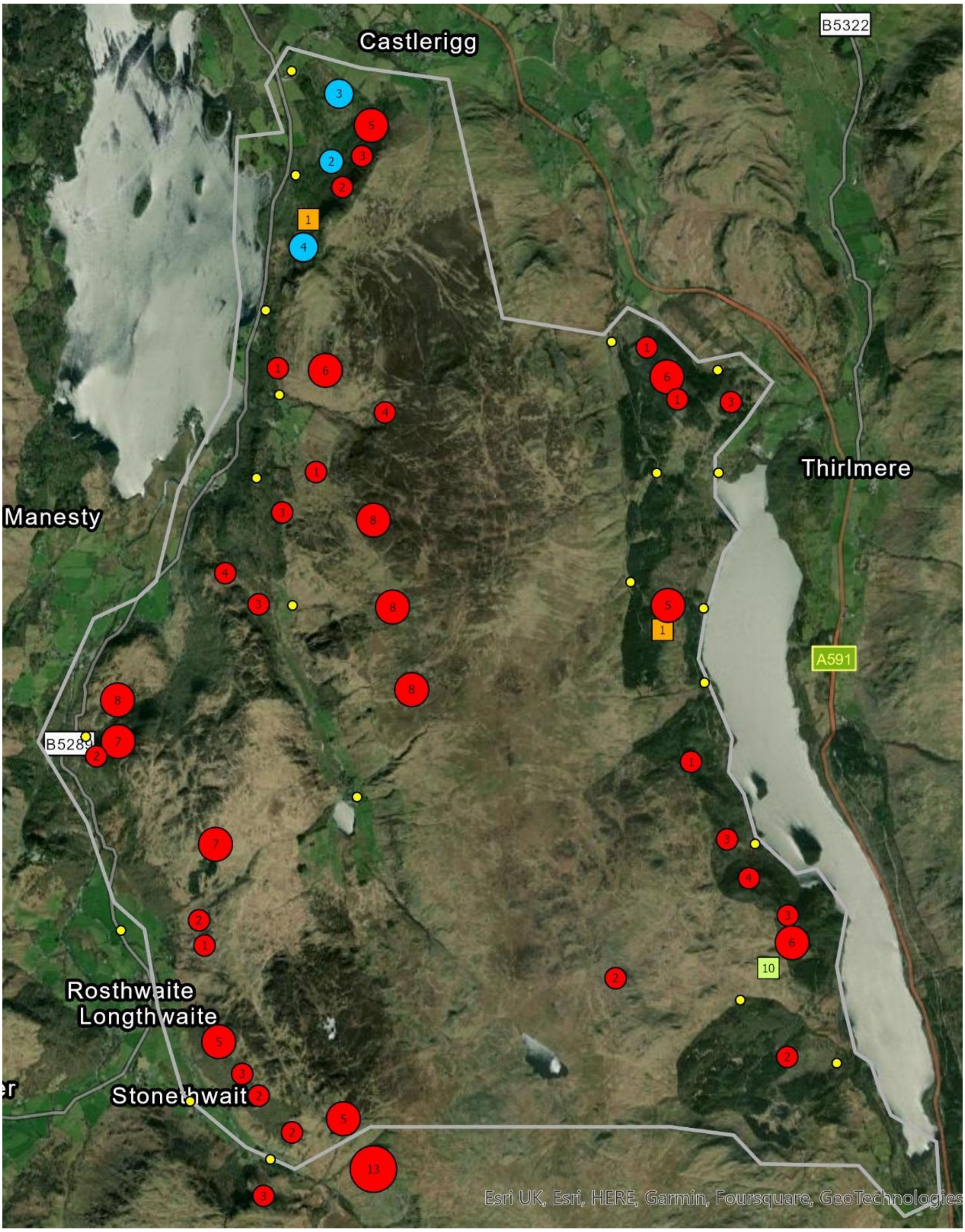

Figure 4.2. Drone survey results map for 2022, created and provided to reproduce in the study by BH Wildlife Consultancy. The grey polygon indicates the approximate survey area boundary; red circles indicate red deer group detections (with the size of each circle and inclusive numbering indicating number of deer); the blue circles indicate roe deer, *Capreolus capreolus*, group detections (with the size of each circle and inclusive numbering indicating number of deer); the green boxes indicate livestock presence; orange boxes were observations of red fox, *Vulpes vulpes*; the yellow circles indicate the drone launch locations, which were consistent in 2022 and 2023.

4.2.e. Line-transect distance sampling

This study used transects due to the varying nature of the Borrowdale and Thirlmere landscape. Between March and May 2022, this study used 14 line-transects between 0.5-1.5km in distance, placed at random to the study area to ensure a representative and proportionate coverage of the area (Figure 4.3). To reduce double counting of detections, the start and end of each transect were at least 0.75km apart, with a minimum of 45 minutes between ending one transect and beginning the next. Further, the groups detected were tracked where possible between transects to reduce double counting deer pushed from one transect to another.

Consistent with the findings of Logan et al. (2019), a thermal imager was used to improve detection probability (Pulsar Helion 2 Thermal Monocular; Pulsar, Worcester, UK; www.pulsaruk.com/) and binoculars to ensure positive identification (Swarowski EL 10x42 binoculars; Swarovski, Absam, Austria; www.swarovskioptik.com). Further, a rangefinder was used to measure the distance of detected animals from the transect (Vortex Ranger 1800; Barneveld, USA; vortexoptics.com), and a compass to measure the angle of detected animals from the observer on the transect. Combining the distance and angle from observer enabled the calculation, using trigonometric functions, of perpendicular distance of detections from the transect. Finally, a GPS unit was used to follow the transect paths (Garmin eTrex 10; Olathe, USA; www.garmin.com).

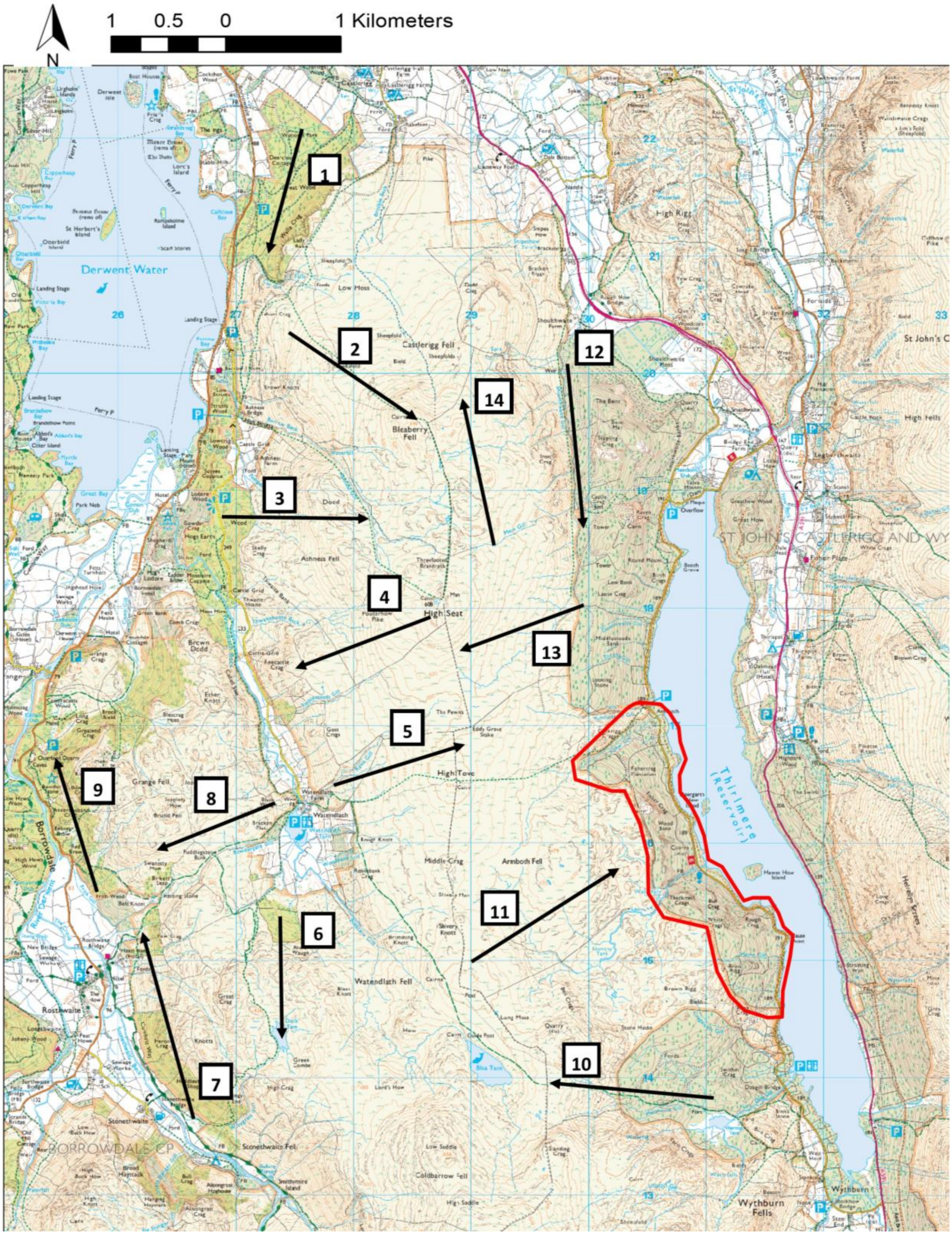

Figure 4.3. Transect map for distance sampling transects in Borrowdale and Thirlmere. Black arrows indicate the approximate location, distance and direction of the transect (taken from beginning and end transect coordinates), number indicate which transect the arrow relates to, and the red polygon indicates an area that could not be surveyed due to safety reasons. The Map was developed from open access OS Maps at 1:50000 ratio.

4.2.e.1. **Distance Sampling Analysis**

Data for distance sampling were analysed using Distance 8.0 (Thomas et al., 2010). The data were left truncated to meet the assumption of perfect detection on the transect line; only four deer groups detected occurred <50m of the transect and there were zero detections on the transect. The data were also right truncated at 600m due to a substantial increase in detections at 650m and 700m from the transect by a consistently located group of deer across a gap between two hillside peaks. A Uniform model with Cosine adjustment was selected after comparing AIC scores of multiple available models (Table 4.1).

4.2.f. Trail cameras

The study site was divided into a grid of 1km^2 cells, delineated using the Ordinance Survey British National Grid to enable even coverage and division of the landscape between the different ecotopes. One camera was deployed per 1km^2 grid to survey the 40km^2 landscape. Cameras were deployed over 86, 74, and 71 days in 2022, 2023 and 2024 respectively to capture a representative sample (defined as a subset of a population that accurately reflects the characteristics, variations, and proportions of the whole population) of the relative activity of red deer across the landscape using fixed camera placements. There were two distributions of the cameras in 2022 and 2023, at 20 cameras per distribution and a movement of cameras half-way through deployments, and 40 cameras deployed in 2024 (more cameras became available to the study). The woodland and open hill ecotopes were given proportional coverage across all distributions (Figure 4.4).

This study used Bushnell Prime 24MP trail cameras (Bushnell, Kansas, USA; www.bushnell.com), placed approximately 1m above ground to enable detection and identification (Rowcliffe et al., 2008) of the red deer. Cameras were placed facing between northwest and north for consistency and reducing false triggers from the Sun on camera motion sensors (consistent with Logan, 2018). All cameras were placed 0.75-1m above the ground, angled between 80-110^o to the ground to ensure an effective field of view for the cameras. In the open hill habitats, camera placement locations were determined from a combination of site visits and local knowledge to find appropriate tree/property boundaries to attach cameras to. In woodland habitats, the cameras were located within 15m of pre-selected GPS coordinates with at least 25m of open view in front of the camera to prevent natural features (e.g. trees) from obstructing detections. Each camera had a 2-minute break between detection events to reduce double counting, with 5 photographs taken at each detection event to enable species identification.

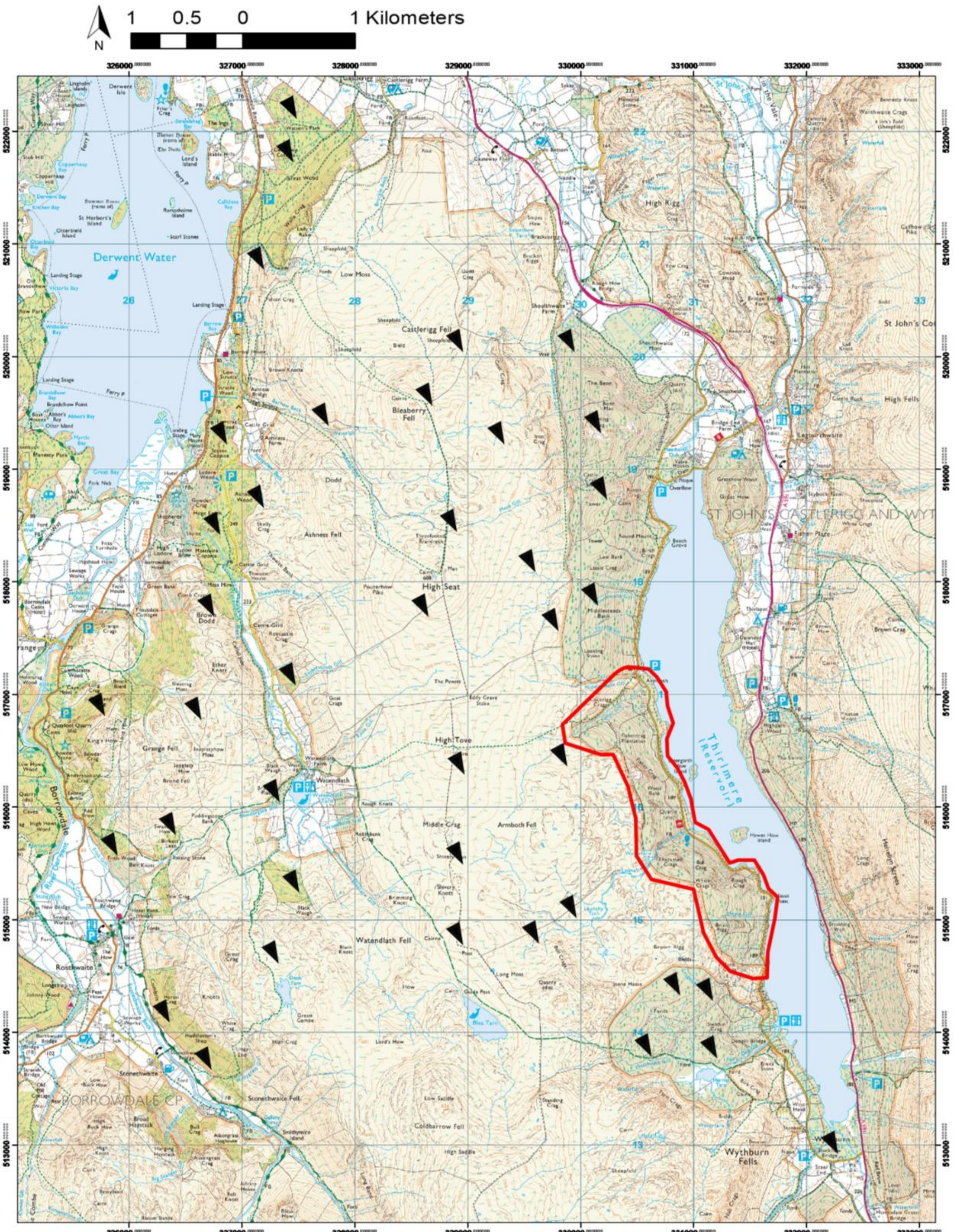

Figure 4.4. Trail camera placement map in Borrowdale and Thirlmere from 2022-2024. Black polygons indicate the approximate location, distance and direction of the transect (taken from coordinates at each camera placement). The red polygon indicates an area that could not be surveyed due to safety reasons. The Map was developed from open access OS Maps at 1:50000 ratio.

4.2.f.1. **Random Encounter Model**

Data were analysed using Random Encounter Model [REM] *sensu* Rowcliffe et al. (2008) (Eqn. A) in Microsoft Excel. REM estimates population density from random encounters of moving animals with static cameras and is targeted towards assessing a target population size (Rowcliffe et al., 2008). Recent adaptations to REM, such as Random Encounter and Staying Time (REST), could have been used. REST was designed to synchronously survey a broader number of species more efficiently than REM was capable as a result of requiring species-specific movement parameters (Nakashima et al., 2018). However, REST assumes that animal encounters and the distribution of staying time reflect a single, shared underlying pattern across individuals, and is therefore susceptible to asynchronous behaviours by individuals significantly impacting density estimates (Nakashima et al., 2018), which is likely in highly gregarious deer species. The REST model does not deliberately target for a species-specific survey and has only previously demonstrated effectiveness in woodland environments with species that express low levels of gregarious behaviour (Nakashima et al., 2018). Thus, REST was considered less likely to be effective in this upland landscape with red deer compared to the original REM.

The original REM (Rowcliffe et al., 2008) is a Poisson process whereby animals are assumed to move randomly through space, and the camera samples a fixed detection zone. Therefore, independence from parameters was required for the encounter rate (y/t; Eqn. A) to scale linearly with density. Thus, to meet the mathematical assumptions of the original model, we required independent assessments of the average daily speed of movement in km/day (parameter v; Eqn. A), the average detection radius and arc of the camera (r and θ; Eqn. A) and a representational average of typical group size (the median was used from estimates gathered during the distance sampling method in 2022, and independent repeats of the same transects in 2023-24 up to 40 group detections).

Average daily speed of movement can be calculated using GPS collars, trail camera video footage, or continuous photographic sequences from trail cameras. However, using the trail cameras deployed for data collection to also estimate average daily speed of movement would violate the assumption of independently assessed parameters. Due to the time required to navigate the study site, the large number of cameras deployed, and the length of time each camera remained in the field, researchers were unable to replace batteries and SD cards at regular intervals. As a result, collecting consistent video footage for estimating movement speed was not feasible. Therefore, average daily speed of movement was calculated from GPS data from Pépin et al. (2004) for red deer in a comparable woodland-mountainous environment. The value was used with the caveat that it may not accurately reflect the movement speed of the deer population in this study, but it provided and consistent and independent estimate.

The radius (*r*) of the camera detection zone was assessed through four repeat trials to five cameras (25% of available cameras) in 2022, adapted from published procedures (e.g. Pettigrew et al., 2021; Rowcliffe et al., 2008). Through these trials, held at Keswick Rugby Club, Cumbria, UK, the cameras were attached to goal posts, whilst adult male rugby players crossed the camera detection zone perpendicularly. Detection distance (r) was recorded based on the mean furthest distance for detection, and the detection arc (θ) of the camera detection zone was estimated by the mean of the widest detection from the cameras. Although calibration in all habitat types and locations may be considered "ideal", the time taken to achieve this is impractical when deploying numerous cameras over a large site with difficult terrain to manoeuvre and would have significantly impacted other fieldwork needs. The use of mean parameters of camera models has been previously accepted in REM studies (e.g. Miura et al., 2022) as the model estimates a mean density over the areas sampled, and were considered an acceptable adjustment to account for fieldwork requirements. This was not repeated later in the study as the model of camera did not change. To be included as an encounter event (parameter y), the image needed to be positively identified as a red deer.

Eqn. A. Random Encounter Model *sensu* Rowcliffe et al. (2008):

$$D = \frac{y}{t}\frac{\pi}{vr(2+\theta)}$$

Where:

D = Density (individuals km^{-2}),

y = camera trapping events,

t = Time of camera trapping effort (in days),

v = Average daily speed of movement (km/day),

r = Effective detection distance of the camera (km),

θ = Effective Angle of the camera detection zone (radians).

4.2.f.2. **Estimating uncertainty in REM**

Variation of the camera trapping data was estimated by the standard deviation of bootstrap randomised resampling (Jourdain et al., 2020) of the camera trapping events (parameter y; Eqn A). The data were resampled 1000 times to calculate the mean and standard deviation. This analysis was performed in Microsoft Excel.

4.2.f.3. **Estimating annual change**

Annual change to abundance from REM measurements was analysed using the estimate of REM (mean and 95% confidence interval). Standard error was calculated from the 95% confidence intervals (Eqn. B.) and were used to generate inverse variance weights. A weighted least squares regression of abundance on Year (coded 0-2) was fitted to test for linear trend. Pairwise year-to-year differences were evaluated using z-tests based on the summed variances of each parameter estimates. To check robustness of variance assumptions, the trend analysis was repeated on log-transformed abundance using standard error from log-transformed confidence interval bounds. These analyses were completed using R-Studio 2024.12.1 with base R.

Eqn. B. Standard Error from 95% Confidence Intervals:

$SE = (U - L)/(2 \times 1.96)$

Where:

SE = Standard Error

U = Upper 95% Confidence Interval

L – Lower 95% Confidence Interval

4.2.f.4. **Estimating sensitivity of REM**

Sensitivity of REM to uncertainty in each parameter was tested by varying the parameters outside our control (y, v, r, and θ; Eqn A). Time was not included as it is under fixed control. The direct effects of parameters under our control (y, v, r, and θ) were tested through individually varying the parameters by ± 5%, 10%, 25%, and 50% of their original value and comparing change to density estimates. This analysis was completed using Microsoft Excel.

4.3. Results

4.3.a. On-foot census counts

For the census counts in September 2021 and 2022, there were between 25-30 observers used across four hours of surveying on open hill and woodland ecotopes. There were reported, after removal of known double counts, 143 and 139 red deer in 2021 and 2022 respectively. Surveyors inconsistently recorded between number of deer per group detected, and a tally of the total number of deer over the whole survey, therefore average group sizes could not be extracted.

4.3.b. Drone survey census counts

In December 2022, there were 21 drone launches, during which 38 red deer groups were detected. The average red deer group size at each detection was a median of 3 (1-9; 95th percentile range), with a total of 154 red deer counted. In December 2023, there were 22 drone launches detecting 63 red deer groups. The average red deer group size at each detection was a median of 3 (1-13.85; 95th percentile range), with a total of 246 red deer counted. All drone surveys were conducted by a single pilot over 12 hours, divided between two morning surveys from 6:30am – 12:30pm.

4.3.c. Line-transect distance sampling

In 2022, a total of 115.5km of transects were surveyed across 28 surveying days, with 7 repeats of 14 transects. There were 755 individual red deer detected across 80 group detections, with a median group size of 5 (2-33; 95th percentile range). The data were left truncated by 50m due to few detections <50m from the transect, and zero detections on the transect, and right truncated from 600m due to an increase in detections of deer from 650m+ (Figure 4.5); right truncation was attributed to consistent behaviour of deer groups between hill tops in this terrain. The effective strip width (ESW) was 291.82m (Table 4.1). The model selected was a uniform model with cosine adjustment (Table 4.1) and was decided by comparing AIC scores. From this model, the average abundance of red deer was estimated at 279 individuals (7km^{-2}), with upper and lower confidence limits of 145-540 red deer (Table 4.1).

Table 4.1. Model selection and outputs for analysing Distance Sampling data in 2022.

Model	AIC	ESW	X^2 Goodness of Fit	D (LCL-UCL)	N (LCL-UCL)	%CV	P (LCL-UCL)
Uniform	**667.31**	**291.82**	**0.404**	**7 (3.62-13.54)**	**279 (145-540)**	**32.9**	**0.48 (0.41-0.57)**
Default	668.42	300.34	0.393	6.8 (3.44-13.46)	271 (137-537)	34.3	0.5 (0.38-0.65)
Negative – Exponential	669.31	291.77	0.306	7 (3.62-13.54)	279 (145-540)	32.9	0.48 (0.41-0.57)
Half-Normal	668.42	300.34	0.393	6.8 (3.44-13.46)	271 (137-537)	34.3	0.5 (0.38-0.65)
Hazard-Rate	668.43	364.56	0.549	5.4 (2.81-10.45)	216 (112-417)	32.8	0.6 (0.49-0.75)

Notes: AIC = Akaike's Information Criterion; ESW = Effective Strip Width; D = Density; N = Population abundance; %CV = Percentage Coefficient of Variation; P = Detection probability; LCL = Lower confidence limit; UCL = Upper confidence limit. This study used the Uniform model with Cosine adjustment (first row).

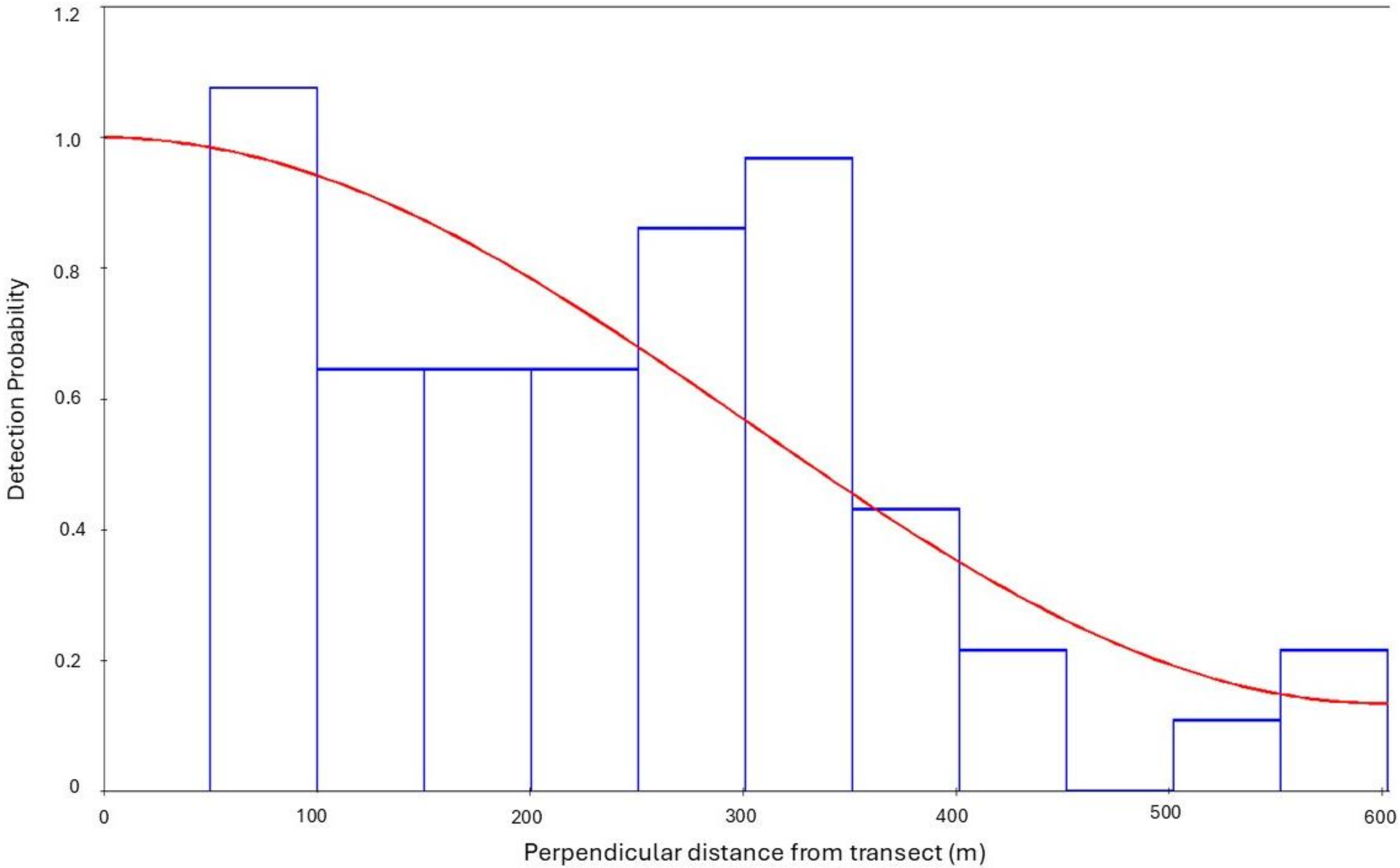

Figure 4.5. Total observed perpendicular distances (histogram) and detection probability (line graph) of red deer, with a Uniform model and Cosine adjustment, left truncated at 50m and right truncated at 600m.

4.3.d. Trail cameras

Across the three years, there were 40 camera placements used. For 2022, 2023, and 2024, trail camera moves required six days of effort. In 2022, 1510 camera trapping days were recorded with 377 red deer detection events, in 2023 there were 1432 camera trapping days with 209 red deer detection events recorded, and in 2024 there were 2672 camera trapping days with 407 red deer detection events recorded.

In 2022 average group size was independently calculated as a byproduct of data collection for distance sampling as a median of 5 (2-33; 95th percentile range). In 2023 and 2024, 4 repeats of the distance sampling transects were used to detect deer groups, with a median of 5 (1 – 28; 95th percentile range) deer per group in 2023, and a median of 5 (2 – 30; 95th percentile range) deer per group in 2024. Variation in the independently estimated mechanical parameters (the detection zone of the cameras) was low from twenty measurements.

The average abundance of red deer was 278 (6.97km^{-2}) in 2022, 260 (6.51km^{-2}) in 2023, and 247 (6.2km^{-2}) in 2024 (Table 4.2). The range of abundance (95% confidence interval) for the red deer (Table 4.2) was calculated from the standard error of the mean for REM.

Table 4.2. Population size estimates from REM from 2022 – 2024.

Population Estimate Parameters	2022	2023	2024
Average density km^{-2}	6.97	6.51	6.20
Standard deviation	1.51	1.22	1.23
%CV	21.67	18.74	19.84
Mean abundance	278	260	247
95% Confidence Interval	259 - 298	245 - 275	232 - 262

From the REM abundance estimates, 95% confidence intervals were used to derive standard errors and conduct inverse-variance weighted analyses. The estimated annual change in abundance was centred on a decline, with the confidence interval spanning moderate decreases to weak increases of abundance per year ($\bar{x}$ = -15.24 (95% CI: -35.12 to 4.63), p = 0.065). Pairwise comparisons indicated that the estimated change in population size between 2022 and 2024 was consistently negative, with the 95% confidence interval excluding zero (-55.6 to -6.4; z = -2.47; p = 0.0135, Figure 4.6 the corresponding confidence intervals for changes between 2022–2023 (−42.6 to 6.6; z = −1.43; p = 0.152) and 2023–2024 (−34.2 to 8.2; z = −1.20; p = 0.230) included zero, indicating substantial uncertainty about the direction of change over these shorter intervals (Figure 4.6). These findings were supported by variance-stabilised sensitivity on the log-scale, which estimated a multiplicative change of 0.943 per year (95% CI: 0.900 to 0.988), corresponding to an annual decline of approximately 5.7% between 2022-2024. The confidence interval indicates that plausible rates of decline ranged from approximately 1.2% to 10.0% per year.

The impact from the camera trapping rate (parameter y) is linear to the density of deer estimated, with reductions in y leading to reduced density and vice versa (Figure 4.7). The average daily speed of movement, and the detection range of the cameras (parameters v and r) have an equal impact to density estimates in REM (Figure 4.7). Further, the changes to v and r past 10% had comparatively larger impacts to density estimates than changes up to 10% (Figure 4.7). The detection angle of the trail cameras had the least impact to density estimates using REM (Figure 4.7).

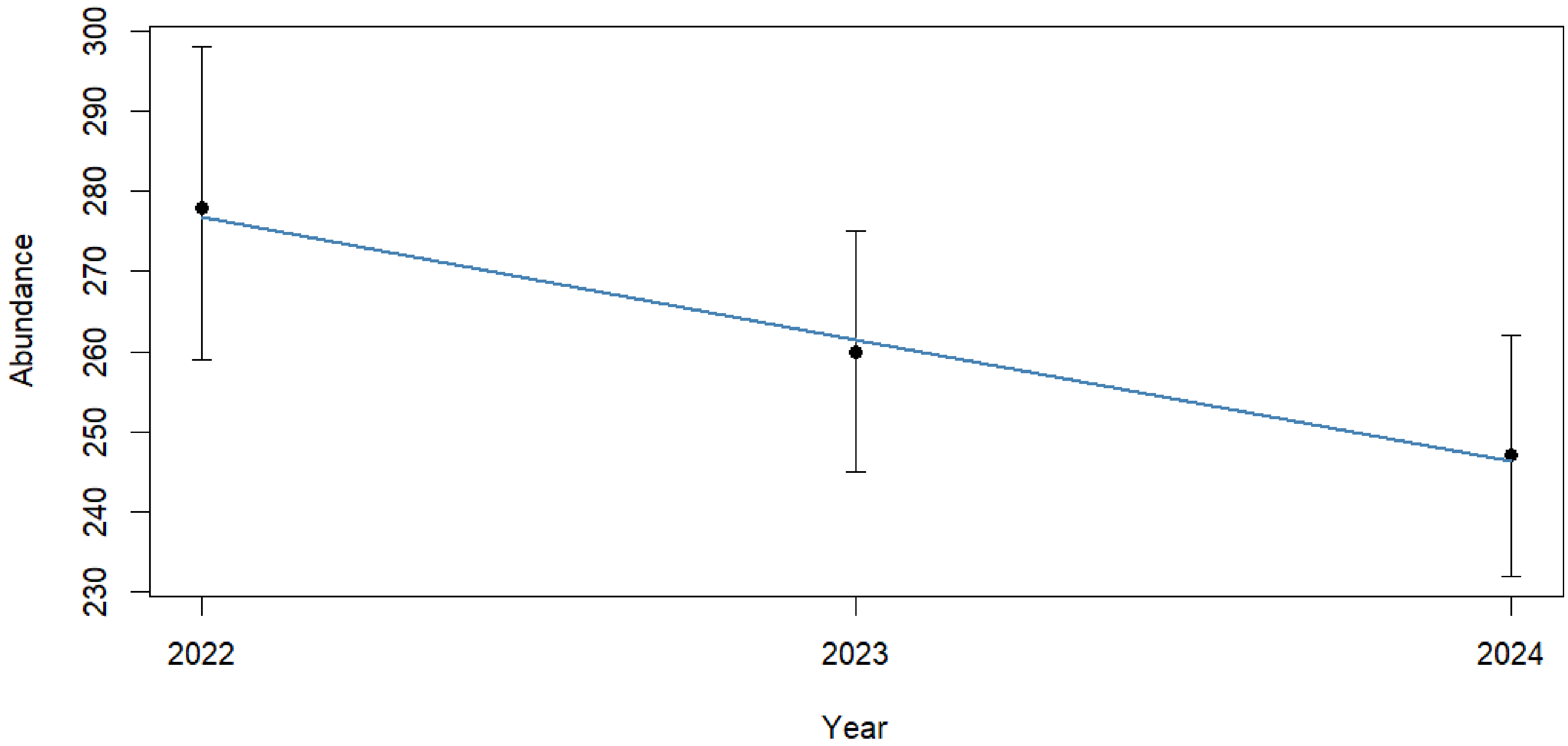

Figure 4.6. Decline in population of red deer in Borrowdale and Thirlmere over the three-year study period using Random Encounter Model. Black circles represent average annual abundance estimates, whiskers show the corresponding 95% confidence intervals, and the blue line illustrates the fitted negative regression trend across the three-years.

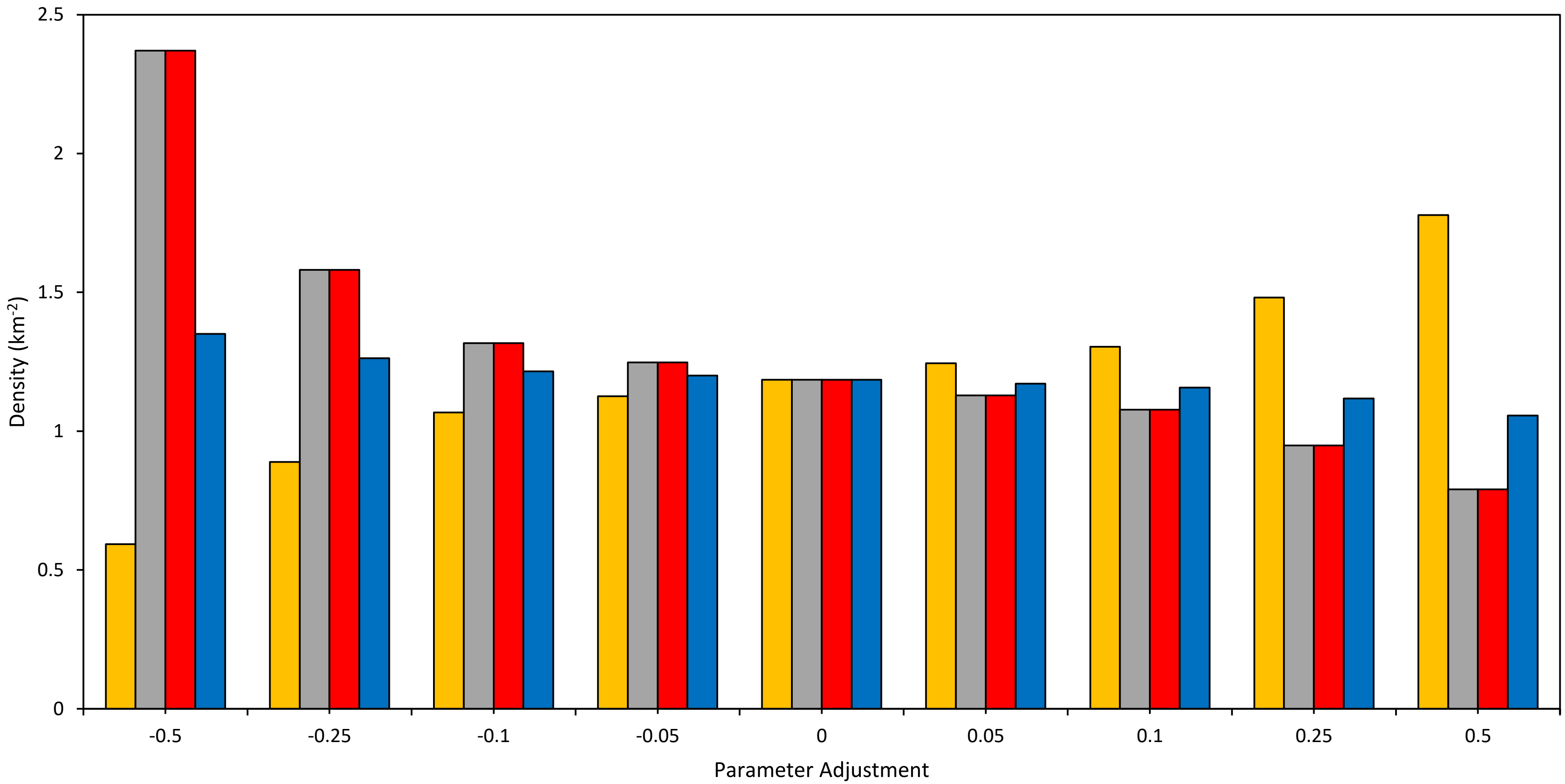

Figure 4.7. Sensitivity of parameters to estimated density from Random Encounter Model. The original parameter values were varied between ± 0.05, 0.1, 0.25, and 0.5 of their original value. Y (yellow) = camera trapping events; v (grey) = average daily speed of movement (km/day); r (red) = detection range of the cameras (km); θ (blue) = angle of camera detection (radians).

4.3.e. Performance of population estimate methods

The correspondence of population estimates for all methods were compared by the difference between the mean values of sampling methods (REM and Distance Sampling) and the mean of REM with the minimum estimated population size of the census methods (Table 4.3). The precision of estimates from sampling methods (%CV) and the effort to collect data in days are also compared below.

REM and Distance Sampling had a close correspondence, with a difference of +1 from mean abundance estimates (Figure 4.8). Distance Sampling had a lower precision (32.9%CV) than REM (21.67%CV), while the effort to complete REM was lower (6 days) than Distance Sampling (28 days) (Table 4.3).

The on-foot census did not correspond closely with the REM in year 1 or year 2, with a difference of -135 and -121 respectively (Figure 4.8). The on-foot census counts were considered a significant underestimate of the population, with the culling effort in the landscape exceeding these minimum population sizes in both years. Further, simple population models demonstrate a negative recruitment using this population data and cull effort, and a population extinction by the end of the second year in both instances. The total survey effort for on foot census was approximately 5 days of effort each year, which is comparable to the effort for the camera trapping for REM (Table 4.3).

The drone census methods corresponded inconsistently with the mean REM abundance in year 2 and 3, with a difference of -106 and -1 respectively (Figure 4.8). The effort to complete drone census was 0.5 days, which was lower than REM and on-foot census (Table 4.3).

Table 4.3. Comparators of population estimate methods

Year	Method	Effort (days)	Precision (%CV)	Difference between estimates
1	Census	5	N/A	-135
	Distance Sampling	28	32.9	+1
	REM	6	21.67	0
2	Census	5	N/A	-121
	Drone	0.5	N/A	-106
	REM	6	18.74	0
3	Drone	0.5	N/A	-1
	REM	6	19.84	0

Notes: Difference between estimates refers to difference between the means for REM and Distance Sampling, and the means for REM and minimum population estimate for Census and Drone. Census effort is based on 30 surveyors at 4 hours per survey. Drone effort is two 6-hour morning surveys.

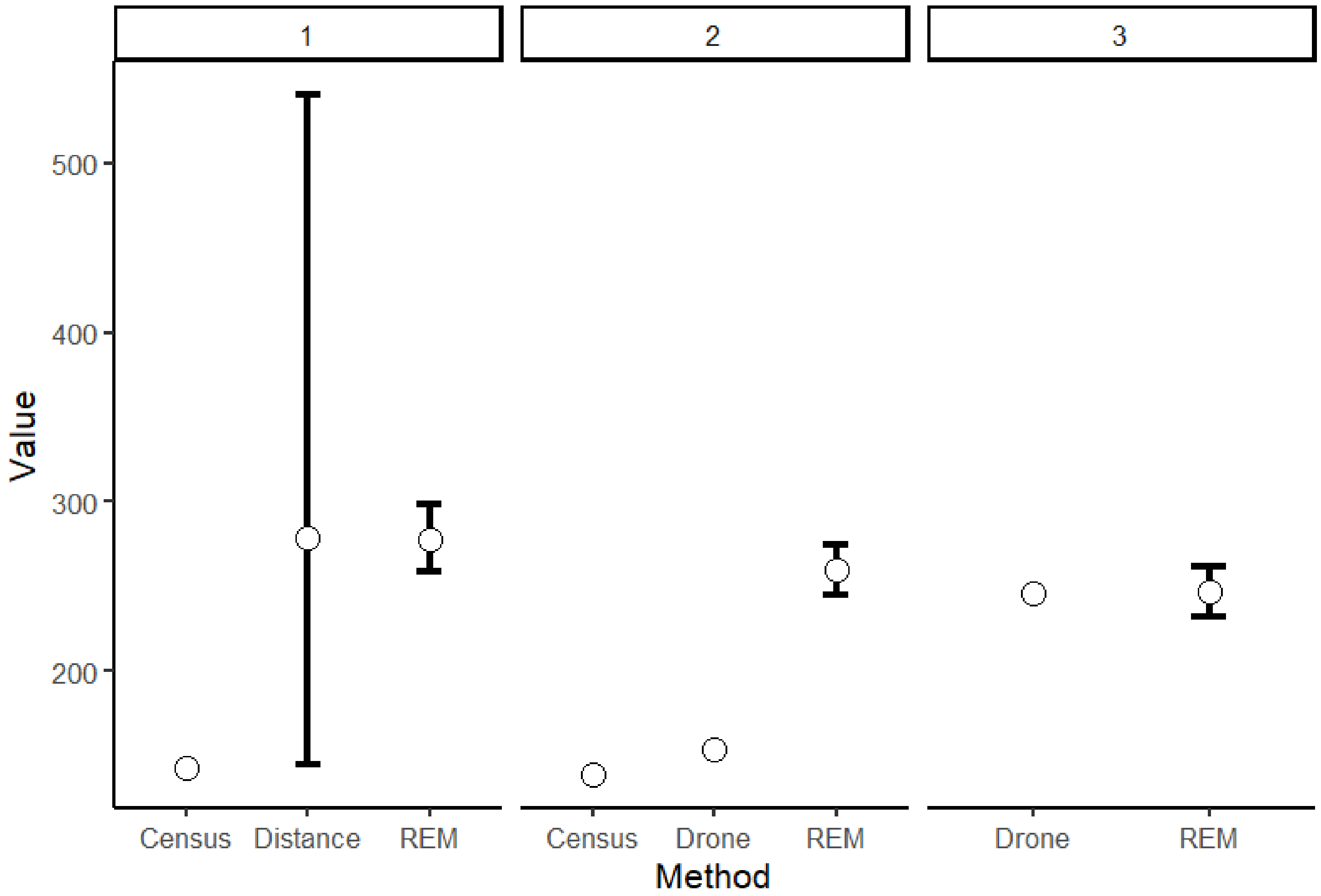

Figure 4.8. Comparison of population estimates over the three years surveyed. Open circles are the only (census) or average (sampling) population estimate available, with range bars demonstrating the upper and lower population estimates from sampling. X-axis is the method used: Census = on-foot census count; Distance = distance sampling; REM = Random Encounter Model; Drone = drone census. Y-axis is the population abundance.

4.4. Discussion

The population estimates from this work were taken from different months each year, with camera trapping and distance sampling methods performed between March and May 2022-24, on-foot census in September 2021-22, and drone census in December 2022-23. A direct comparison between distance sampling and REM in 2022 is a valid comparison as they were performed at the same time. But attempts to directly compare estimates between the census methods, and comparing between census methods and sampling, should be made with caution. This is due to the legal culling season of female deer between 1st November and 31st March in the Deer Act 1991, and parturition in June-July (Mattioli et al., 2022). Each such event significantly alters the population size between the population abundance estimate methods (i.e. September for on foot census, December for drone census, and March-May for sampling). Further, the census methods were unable to estimate population size, instead quantifying minimum population size that underestimated the deer population in each estimate comparative to sampling methods (apart from the 2023 drone census). There will be natural migrations and mortality that may impact population estimates, but these are not expected to change that population size significantly. The sampling methods will be less prone to impact from migration compared to census as they sample temporally (REM and Distance Sampling) and as a product of relative activity (REM).

On foot census counts were low effort to perform, with no analyses of data required, making a simple method that may be appealing to practitioners. However, the data as collected and reported did not provide the capability to estimate a range of population size that could be used to inform management actions (Forsyth et al., 2022) and account for statistical uncertainty in population harvesting (see Ward et al., 2020). This is demonstrated with the disparity between census count and recorded culls of 143 counted and 155 culled in 2021-22, and 139 counted and 119 culled in 2022-23. Such population data could not have been strictly adhered to by the deer managers to inform decision-making given culling more than 100% of the estimated population in 2021-22. Extirpation of this deer population did not occur, suggesting a significant underestimation of the deer population size from census counts. Yet, it would be possible to extrapolate and model a range of population information with a more formalised approach to the on-foot census counts. By capturing the variability of personnel, detection equipment available per person, and the time and habitats surveyed, such variability could function as a proxy for uncertainty that can be modelled from bootstrap resampling or monte carlo simulations. This may be a more complex analytical process than currently performed, but should practitioners capture this data, it could be outsourced for analysis for a more statistically reliable on-foot census count to support management decision-making.

The drone censuses were low effort, although incur a set cost per survey, with minimal interrogation of the permanent data (images and coordinates of detections) required. The drone had a higher detection rate compared to on foot census but provided estimates with varying correspondence to REM of 106 fewer deer in year 2, and 1 less deer in year 3 (Table 4.3). The population estimate from drones in year 3 took place after the 2023 parturition, and five weeks into the 2023-24 culling season (December 2023), whilst REM for year 3 took place in March-May 2024. The closeness of the estimates, despite a cull occurring between them, would suggest that one or both of the methods had low accuracy. Alternative explanations could include temporal variability in population estimates leading to poor comparisons (e.g. Smart et al., 2004) or a high immigration rate to the study area. To answer this fully, an evaluation of available data on population estimates and culling using population models may produce estimated migration rates for comparisons. Further, the accuracy of REM should be tested.

REM is a well-studied and evaluated sampling method, as is distance sampling. The average population estimate using distance sampling in this work was different to REM by 1 deer. However, with high variance in population estimates negatively impacting management decision-making (Ward et al., 2020) and low precision from distance sampling estimates in this work (and the effort involved to collect and analyse data) REM would be the more precise sampling method, with lower effort to use, from this work.

Palencia et al. (2022) concluded REM to be an effective and “reliable” analytical tool when the parameters are “adequately estimated”. Sensitivity analyses showed that the detection range of the cameras and the average daily speed of movement were the most influential parameters on REM (Figure 4.7), making their accurate estimation critical. For the detection range of the cameras, there are two key methods currently used to estimate this. The first is to perform systematic ‘walk tests’ in front of the cameras, by walking perpendicular from varying distances away from the camera to identify the average angle and distance of detection (e.g. Pettigrew et al., 2021; Rowcliffe et al., 2008), as in this work. Alternatively, some applications of REM estimate detection distance and angle trigonometrically from camera images, using specific markers placed in situ as reference points, and calculate the average range and angle as the effective detection of the cameras (e.g. Pfeffer et al., 2018; Wohlfahrt et al., 2025). In each instance, these were an estimation of average performance of the cameras across the entirety of the study site being estimated, which would account for some variation across different habitats.

Typically woodlands are likely to be more obstructive to effective detection zones of trail cameras, due to vegetation structures, than open range habitats, which risks impacting trail camera performance. To counter that in this study, trail cameras were placed with a minimum 25m clearance for unobstructed fields of view. As such, it was considered that walk-testing a subset of cameras in a clear setting would constitute the “adequately estimated” parameters described by Palencia et al. (2022). This aligns with decisions in other studies (e.g. Miura et al., 2022; Pettigrew et al., 2021), where physical constraints (e.g. dense vegetation, difficult access, challenging terrain, and a need to maintain operational efficiency for batteries and SD-cards), limited the feasibility of walk-tests at every camera location. Further, Schaus et al. (2020) concluded that using averaged parameters across large-scale sites did not significantly bias population estimates when compared directly with camera placement-specific parameters. Borrowdale and Thirlmere, at $40km^2$, sits within the spatial scale for which parameter averaging was demonstrated as appropriate (Schaus et al., 2020). Consequently, using representative, rather than site-specific walk-testing, was judged to provide a balance between methodological rigour and logistical practicality.

To estimate the average daily speed of movement, two strategies are commonly used for REM project: attaching GPS collars to a sub-sample of deer (e.g. Graves et al., 2022; Pépin et al., 2004; Silovský, 2024; Yuan et al., 2019); or estimating speed using trail cameras (e.g. Pfeffer et al., 2018; Rowcliffe et al., 2016; Wohlfahrt et al., 2025). GPS collars provide precise movement data but are costly and require licensing in England for capturing deer (Deer Act 1991), and licensing under the Animals (Scientific Procedures) Act 1986 if immobilising using tranquiliser darts for scientific research. Successfully attaching collars to deer in the UK have rarely been successful;

for example, Smith (2025) only attached one collar to a single red deer throughout a multi-year project. The use of trail cameras, as with camera effective detection parameters, requires markers to be left to measure distances travelled (Nakashima et al., 2018; Pfeffer et al., 2018; Wohlfahrt et al., 2025).

In this work, the terrain was challenging, with difficult access that impacted time and capabilities of moving cameras/changing batteries and SD cards without assistance. As such, maintaining operational efficiency for the batteries and SD cards of the cameras was a priority, impacting the camera set up and reducing the opportunity to use cameras for estimating average speed of movement, or the detection range of the cameras. Further, the study site contained numerous conservation designated sites, where a key agreement with the landowners was not to make any temporary or permanent changes to the environment beyond the placement of trail cameras – impeding the ability to place markers in the background of the camera view used to reference distance for both camera parameters and average movement speed. As such, the average daily speed of movement could not be calculated using the trail cameras, nor from GPS collars, and therefore was taken from literature from a red deer population in a mountainous, forested, landscape in Europe (Pépin et al., 2004). This figure of movement speed cannot be considered accurate for this population but provided a consistent parameter that could be quantified at a later date as part of an updated study, consistent with the rationale of Cusack et al. (2015) and Popova et al. (2019). In the meantime, the figures produced cannot be considered accurate, but the consistencies of parameter values increase confidence in the validity of the trend of marginally significant population reduction over the three years.

The REM analysis detected a marginally significant decline in red deer abundance between 2022 and 2024, corresponding to an estimated annual reduction of approximately 5.7%. This decline was not expressed as significant year-on-year change, but rather as a cumulative difference across the full study period, indicating a gradual and sustained population response rather than episodic fluctuation. This was identified from pairwise z-tests, demonstrating that average abundance was not significantly different from year-on-year comparison (2022-23 / 2023-24), yet there was a significant decline when evaluating between 2022-24. Such a pattern was consistent with the “modest population reduction” target from landscape-scale management interventions, which was intended to stabilise or slowly reduce abundance rather than induce rapid declines (Pers Comm, Harrington, T). The marginal statistical support for this trend is therefore likely attributable to the short temporal window combined with small annual effect sizes, rather than an absence of demographic signal, and should be interpreted accordingly. Importantly, although this detected decline aligns with stated management objectives, the limited time series precludes strong conclusions about longer-term population trajectories.

Importantly, consistent methodology is widely recognised as sufficient for reliable inference on relative population change, even where point estimates may be biased, whereas inconsistency can obscure biological signals entirely (Kéry & Schmidt, 2008; Nichols & Williams, 2006). However, because the average daily speed of movement parameter was applied consistently across all years, the relative changes in abundance were likely robust (Kéry & Schmidt, 2008; Nichols & Williams, 2006). Accordingly, the REM results are best interpreted as evidence of a plausible and management-consistent decline rather than precise quantification of population size, underscoring both the utility of REM for detecting early population responses and the need for population-specific parameter optimisation and longer monitoring periods where annual abundance estimation is a primary objective.

REM has previously demonstrated itself to be a practical way of analysing the abundance/density of cryptic species from remote sensing data across landscapes of varying sizes. It has been increasingly used for estimating the density of species of conservation importance to assist with management and intervention planning (e.g. Ahmad et al., 2024; Gray et al., 2020; Rademaker et al., 2016). The performance of the model, from the original REM (Rowcliffe et al., 2008) to more recent derivations (e.g. REST, Nakashima et al., 2018), was still critically dependent upon camera parameters, average daily speed of movement of the species being surveyed (REM only), and unbiased camera placements (Cusack et al., 2015; Janječić et al., 2025; Rademaker et al., 2016; Rahman et al., 2017). Although many REM projects had used camera derived estimates of speed of movement and effective detection parameters (e.g. Pfeffer et al., 2018; Rowcliffe et al., 2016; Wohlfahrt et al., 2025), the prevailing view was that a minimum of 50–70 detections to reliably estimate these parameters (Wiegers et al., 2026), which not all projects would be able to achieve. Further, estimating these parameters from the cameras violates the Poisson distribution assumption of REM, and therefore risked biasing the estimates produced.

REM studies had increasingly concluded the need for independent verification from a different method to validate the estimates produced (Ahmad et al., 2024; Cusack et al., 2015; Gray et al., 2020; Palencia et al., 2022), where such methods (e.g. dung counts, visual surveys, and distance sampling) would reveal significant method-specific biases when applying REM (Balesteri et al., 2016; Bollen et al., 2023; Kavčić et al., 2021; Soofi et al., 2017). Such a comparison was applied within this work through comparing distance sampling and REM in 2022, whereby there was a near identical mean population density and abundance estimate. Although not fully conclusive, this was considered an indicator that the REM estimates from this work could be close to the true population value.

Importantly, recent adaptations and applications of REM were targeted towards statistical robustness and countering early practical difficulties from estimating camera and biological parameters (e.g. Janječić et al., 2025; Nakashima et al., 2018). However, achieving these improvements was accompanied by increased analytical complexity. Such complexity is likely to place newer iterations of REM beyond the practical capabilities of non-academic practitioners, thereby creating a barrier to the uptake of otherwise effective methods (Buckland et al., 2023). Practitioners typically prioritise approaches that are sufficiently reliable to inform decision-making rather than methodologically optimal, favouring a compromise between accuracy and usability (Buckland et al., 2023). In this context, the original REM, implemented through a simplified analytical workflow (for example, via an accessible Excel-based tool), may be more appealing to practitioners, whereas more recent, complex versions of REM risk being disregarded due to their limited accessibility.

Wildlife management and conservation actions require precise data for effective decision-making. But the need for modelling data to achieve this adds a complexity that likely hinders non-academic practitioners from taking up the use of such methods (consistent with Lorenzini et al., 2022). Low effort sampling methods that can have complexity removed from them, such as user-friendly software to perform data modelling for the user, may improve the uptake. Further, REM may have been the most effective sampling method in this study, but it is clear from other works that method selection should be context dependent based on target species and landscape surveyed (e.g. Ward, 2001; Smart et al., 2004; Amos et al., 2014; Forsyth et al., 2022). To achieve this, Forsyth et al. (2022) concluded from a systematic review that for all methods used in population estimates, the biases of methods need assessing, precision of estimates need evaluating, and both must be reported. Reporting bias is required to ensure that decision-making is not based upon a flawed assumption (Forsyth et al., 2022). Further, reported precision increases the likeliness of selecting methods that facilitate effective decision making that accounts for uncertainty in estimates (see Ward et al., 2020). Further, Forsyth et al. (2022) recommended using methods that increase and estimate detection probability as a method to demonstrate bias in population estimating. Through this, wildlife and conservation managers will be better able to make informed decisions that help to achieve wildlife management and conservation objectives.

4.5. Conclusion

Of the methods used to estimate red deer populations in this study, REM provided the highest comparable precision in estimates, at relatively low effort. But the modelling for REM, especially more modern applications, is not user friendly for practitioners to uptake. If REM data analyses can be simplified towards becoming user-friendly, such as using the original model rather than more recent developments, it could be readily used by practitioners. An estimate of daily speed of movement from this population of red deer is required to be more confident in the estimates from REM. Although the correspondence between the independently analysed averages of REM and distance sampling suggested that REM in 2022 produced a potentially credible estimate, a single instance does not allow this to be concluded. Distance sampling corresponded almost identically to REM but had lower precision than REM and a substantially higher effort to complete. This impeded distance sampling from being from usable in management decision-making when considering uncertainty in population estimates impacting population harvesting.

On foot census estimates from this study consistently underestimated the size of this red deer population and was reported in a way that excluded to ability to model uncertainty. A clearer method with the necessary data may allow for this, but such an analysis would likely be too complex for practitioners to complete. Although drone surveys produced permanent and descriptive population data for scrutiny, it did not demonstrate itself as consistently more reliable than on-foot census or either population sampling methods in this study. As such, the data is currently too inconsistent to make wildlife or conservation management decisions that could achieve established objectives.

Overall, the relative low effort of census counts (whether on foot or drone) will be appealing to practitioners for use, with modelling data outsourced to contractors to complete for them. Yet, if REM could have a more reliable parameter estimation of daily speed of movement, then the use of trail cameras (which are not complex to deploy) and either a simplified method of analysis, or a contractor to analyse the data, would likely provide practitioners a robust estimate of population size that offers the ability to quantify the size of change in population abundance over this study’s surveyed timeframe.

Chapter 5

Evaluating deer impacts on upland habitats

Abstract

Large-bodied herbivores have the capacity to influence ecological processes in ways that can be environmentally beneficial or detrimental, depending on human derived landscape management objectives. Deer offer a good example of this, affecting species richness and habitat structure across diverse ecosystems, including woodlands, and upland heather moorlands and grasslands. Despite their ecological significance, methods for assessing deer impacts to inform management and policy remain limited, often relying on unvalidated indexed methods.

This study evaluated the performance of rapid, index-based methods for assessing deer impacts in northern England, applying the Putman Method to upland habitats and the Deer Activity and Impact (DAI) method to woodlands. Both methods were evaluated against intensively sampled plot-based assessments of impact using Spearman rank correlations. Generalized linear mixed models (GLMMs) were used to evaluate whether temporal changes in impact were detectable over three years. Finally, the relationship between relative activity and impacts were evaluated by Spearman rank (index impact) and Person's (intensive impact) correlations.

Both the Putman Method ($r_s(38) = 0.893$, $p < 0.001$) and DAI ($r_s(18) = 0.809$, $p < 0.05$) strongly and significantly correlated with the independent intensive samples of impact for woodland and upland habitats. No significant changes in impact were identified over the three-year time-scale studied using Putman Method ($\chi^2 = -0.497$, $p = 0.619$) or DAI ($\chi^2 = 0.346$, $p = 0.729$). Further, there was no significant relationship between relative activity of red deer and both intensive and index impact assessments. Validated index-based methods for assessing deer impacts, such as the Putman Method and DAI, provide reliable practitioner-friendly alternatives to intensive surveys. Key indicators such as ling heather, sward height, impact to coppice, and browsed bramble were key indicators driving impact scores, although preferential foraging (e.g. blaeberry) highlighted non-linearity in scaling impacts across vegetation types.

Overall, these findings demonstrate that independently validated index-based habitat impact assessments can reliably estimate realised herbivore impacts at landscape scales. The DAI and Putman Method represent scalable, low-complexity, and practitioner-friendly tools that can improve consistency in monitoring, strengthen evidence-based deer management, and support conservation policy in the UK, with

broader relevance for temperate landscapes in Europe, Asia, Australasia, and North America.

5.1. Introduction

The family Cervidae (deer) is a widely distributed grouping of ungulate species that can have substantial ecological impacts across their extant ranges (Baiser et al., 2008; Baruzzi & Krofel, 2017; Martin et al., 2018; McShea & Rappole, 1992; Müller et al., 2017; O'Neill et al., 2022; Ramirez, 2021; Smit & Putman, 2011). The ecological impacts of deer are a byproduct of a combination of feeding behaviours, trampling, fraying/thrashing of plants with antlers, tree bark damage with tusks, and seed dispersal (Gill, 1992; Reimoser & Putman, 2011; Smit & Putman, 2011; Fattorini et al., 2020; Corlatti & Zachos, 2022). But how impacts occur is dependent upon species specific behavioural ecology: e.g. red deer, *Cervus elaphus*, can heavily browse and bark strip trees (Staines & Welch, 1984; Welch et al., 1987; Cukor et al., 2019; Mattioli et al., 2022); while roe deer, *Capreolus capreolus*, browse new growth from commercially planted forests (Tixier & Duncan, 1996; Tixier et al., 1997; Duncan et al., 1998; Ara et al., 2022), native woodlands, orchards, vineyards, and horticultural crops (Tixier & Duncan, 1996; Tixier et al., 1997; Duncan et al., 1998; Putman & Moore, 1998; Lorenzini et al., 2022). Crucially, overabundant populations of cervids have the capacity to impact the stability of biodiversity across their ranges and significantly alter plant communities (Reimoser & Putman, 2011; Smit & Putman, 2011; O'Neill et al., 2022; Fairfax & Westbrook, 2024).

Deer impacts change ecological conditions and anthropogenic resources, which can be perceived positively or negatively based on human-determined management objectives (Putman, 1987; Reimoser & Putman, 2011; Ramirez, 2021). Deer are capable of positively impacting habitats through grazing and browsing plants, creating areas of bare soil, and seed dispersals from their dung (Gill, 1992; Gill & Beardall, 2001; Reimoser & Putman, 2011; Smit & Putman, 2011; Fattorini et al., 2020; Corlatti

& Zachos, 2022). However, negative deer impacts are predominantly based upon feeding behaviours (e.g. extensive browsing damage forestry or homogenized floral compositions from overgrazing moorlands) and trampling of habitats from the activity of larger gregarious species, such as fallow (*Dama dama*) and red deer (Welch et al., 1987; Reimoser & Gossow, 1996; Putman, 2010; Putman et al., 2011a; Rao, 2017; De Marinis et al., 2022; Mattioli et al., 2022; Gresham et al., 2025; Smith, 2025). Although commonly considered important information, the population sizes of deer alone are inconsistent predictors of negative impacts, rather numerous site-specific abiotic factors (e.g. soil depth, productivity, slope aspect) and the population density affect the impact from deer (Putman et al., 2011a; Moore et al., 2018). Instead, monitoring deer impacts may be more useful than monitoring population size under circumstances where impact reduction is a primary aim of management. Impact data would therefore be used to directly inform management decision-making, with density estimates used to assist in planning and monitoring populations; an approach advocated by Ward et al. (2008) and Putman et al. (2011b).

Deer impact management often comprises strategic actions (e.g. fencing, culling) to achieve established landscape objectives through mitigating perceived negative impacts from deer populations (Apollonio et al., 2010; Morellet et al., 2011; Putman, 1987; Putman, 2024; Ramirez, 2021; Reimoser & Putman, 2011). Across continental Europe, the processes of deer management decision-making differ based on national legislative frameworks. Throughout Europe, the use of deer impact assessments to inform deer impact management actions was not commonplace (Putman, 2011), with only a few examples found with evidence of regular deer impact assessments to inform management, e.g. in Norway (Andersen et al., 2010) and Germany (Wotschikowsky, 2010). Conversely to continental Europe, deer management in the UK has a lengthy history of using deer impact assessments as a part of deer management planning (e.g. Holloway, 1967; Melville et al., 1983; Ward, 2001; Ward et al., 2004; Albon et al., 2007; Logan, 2018). However, deer impact assessments are voluntarily collected in the UK, leading to some deer management actions not being informed using best practice methods.

Thus to improve successful outcomes for UK public conservation objectives, there are multiple Environmental Land Management Schemes (ELMS) used by government agencies to financially incentivise the voluntary participation by landowners. For example, WS1 is an England only ELMS administered by the Forestry Commission (2023) to improve woodland management through agreed deer management actions from those who voluntarily participate in the ELMS, including annual assessments for deer impact. Currently, no such ELMS exists for upland habitats, such as heather moorland. However, heather moorlands have long been considered important habitats to maintain for their assemblage of plant species, role in maintaining peatlands, and animals of conservation importance, including reptiles and birds (e.g. black grouse, *Lyrurus tetrix,* and red grouse, *Lagopus scotica*) (Thompson et al., 1995). Yet, native herbivores in the UK (e.g. red deer, a gregarious mixed-feeding species) inhabit and impact both woodland and heather moorlands in upland regions in the UK (Albon, et al., 2007; Moore et al., 2019; Mattioli et al., 2022).

Such upland habitats in northern England have resident red deer populations, with numerous areas that have legally protected conservation designations (e.g. Special Areas of Conservation (SAC) and Special Protected Area (SPA) sites). The cascading effects of red deer on upland ecosystems, through consumption and trampling of vegetation used by ground-nesting bird species, can undermine conservation objectives and necessitate targeted deer management. For example, heavy browsing of *Calluna vulgaris* by red deer (Riesch et al., 2020) can degrade the heather structure that red grouse use for nesting (Ludwig et al., 2020). As ELMS has been used to promote the completion of deer impact assessments to improve woodland regeneration and deer management efforts, similar ELMS could be used to similar effect for upland habitats. Should an upland ELMS be created, a method to reliably record and monitor deer impacts would be required. Further, with reported and projected growth and expansion of deer population ranges across the UK (see Ward, 2005; Palmer, 2014; Croft et al., 2019), effective and consistent monitoring of deer impacts to both private and public objectives is needed.

Assessing deer impacts is not simple, and there are no clear agreed methods/ protocols of how best to do so. The impacts of deer are context and habitat dependent (Putman, 2011a; Putman, 2024) with multiple methods available to assess them in woodland habitats (e.g. Melville et al., 1983; Wild Deer Best Practice Guidance for Scotland, 2018; Armstrong et al., 2023; Forestry Commission, 2023), or open hill/moorland habitats (e.g. MacDonald et al, 1998; Wild Deer Best Practice Guidance for Scotland, 2018; Moore et al., 2019). Industry practitioners require impact assessment protocols that are not complex, quick to apply, and low in cost (Maffey et al., 2016; Månsson et al., 2023; Palmer et al., 2021). Currently, multiple impact assessment methods exist in GB that do not have consistent protocols to each other for similar habitats. This includes multiple methods for woodland impact assessments using either quadrat or transect based data collection protocols, and index-based impact values.

Currently, in Scotland there are a range of methods available to use to assess herbivore habitat impacts (to encapsulate the impact of wild deer, feral goats and domestic livestock) across open range and woodland habitats (Table 5.1). Conversely, in England, two methods are in regular use for woodland impact assessments only, the Deer Activity and Impact (DAI) method and the deer best practice guidance (woodlands) method (Table 5.1), the latter being used by the Woodland Trust for consistent monitoring across all of their properties throughout Great Britain. The DAI method has been in long term use in parts of England and Wales and has become the preferred method of reporting deer impacts from the voluntary recipients of 'WS1: Deer Control and Management', an incentive-based Environmental Land Management Scheme (ELMS) in England (Forestry Commission, 2023). Despite numerous areas of upland grassland and moorland in England, with populations of large grazing deer species on those lands (e.g. Exmoor, Lake District National Park, Peak District) there are no open hill habitat assessment methods in widespread use.

Table 5.1. Common deer habitat impact assessment methodologies used in the UK.

Method (source)	Habitat Type	**Where?**	**Index-based or percentage estimates of impact**	**Breakdown of key method features**
MacDonald method *(MacDonald et al., 1998a; 1998b)*	Open Range	Scotland	Index	Previous preferred method of NatureScot. This method requires pre-defined plots used at a proportionate scale to the area being surveyed. These plots are pre-determined using GIS. Long-term monitoring requires returning to permanent plots.
Wild Deer Best Practice Guidance *(Wild Deer Best Practice Guidance for Scotland, 2018)*	Open Range	Scotland	Index	Open Hill method was derived from the MacDonald Method. This method requires pre-defined plots used at a proportionate scale to the area being surveyed. These plots are pre-determined using GIS. Long-term monitoring requires returning to 30 permanent $4m^2$ plots.
	Woodland	Scotland and England (Woodland Trust Estate)	Index	Woodland method was developed by Forestry Commission for Scotland and NatureScot. This method can be completed using compartmented transects, fixed-point plots (assessing proportion of trees within a plot showing signs of damage) or individual tree monitoring. The number of plots is dependent on the size of the woodland.

Putman Method *(Pers Comm, Putman, R.; Table E1)*	Open Range	Scotland, Northern England	Index	Simplified from the MacDonald Method. A walkthrough method using 12 random stations (selected by throwing a tennis ball) throughout delineated areas of dominant habitats.
Armstrong Method *(Armstrong et al., 2023)*	Woodland	Scotland	Index	This is a plot-based method developed through support from Scottish Forestry. Through a walkthrough of each woodland, a number of "stops" are used to survey within a plot of an approximately 25m radius. The number of plots is dependent on the size of the woodland being surveyed.
Melville Method *(Melville et al., 1983)*	Woodland (Forestry)	England, Scotland, Wales	Percentage	This is a plot-based method developed for the Forestry Commission. This method uses equidistant plots to perform a nearest neighbour assessment of herbivore impact to trees. The number of plots (minimum 20), distances between plots, and number of trees to survey, is calculated using formulas available in the information document.
Deer Activity and Impact Method *(Forestry Commission, 2023)*	Woodland	England, Wales	Index	Developed by the Deer Initiative partnership and a method used as part of compulsory recording for the WS1 grants scheme in England. A transect method, recording a tally of indicators as a sign frequency per 1km.

Industry practitioners (e.g. deer managers) and landowners, rather than professional ecologists, are the primary user of index-based impact assessment methods, demonstrating the need for low-complexity methodology to improve assessment uptake (Maffey et al., 2016; Månsson et al., 2023; Palmer et al., 2021). The indexed methods commonly used to assess deer impacts in the UK use quantitative measures of qualitative descriptors to ascertain different levels of deer impacts, with modal index values used to describe the whole survey (MacDonald et al., 1998a; 1998b; Wild Deer Best Practice Guidance for Scotland, 2018; Armstrong et al., 2023; Forestry Commission, 2023). Most qualitative descriptors used do not appear to be based on peer-reviewed literature, although some non-peer reviewed literature has been used (e.g. Tabor, 1993; Cooke 2007; 2009), instead relying upon the expertise of the developer of the method. Further, we do not know whether the descriptors used to qualify the low-moderate-high indexes in these impact assessments are mathematically discrete/consistent.

Although the woodland and upland impact assessment methods have not previously been validated, the completion of annual impact assessments is becoming increasingly close to a compulsory requirement of landowners in Scotland. In England the DAI method is used as a standard approach, through contractual agreement, by recipients of the WS1 ELMS to report the outcomes of agreed deer management actions (Forestry Commission, 2025). Moore et al. (2018) also identified that although many of the impact assessments methods used in Scotland provided an acceptable description of impact across landscapes, all of them overestimated damage from deer. Without a validation of these methods, which are currently used to inform deer management decision-making, there cannot be confidence in the results produced. Validating indexed assessments of impact by evaluating them against intensively sampled assessments is therefore needed to build confidence in, or offer improvements to, the current methods (consistent with Moriarty et al., 2018).

We aimed to evaluate the performance, based on relative accuracy and precision, of impact index methods in woodland and upland environments. Therefore, we sought to compare the performance of two index methods against more intensive, proportioned, assessments of impact. Well-performing index-based impact assessments will have a very strong positive, and highly significant correlation with the outcomes of intensive impact assessments. A highly significant correlation would indicate accuracy (the closeness of estimates to the true value) and precision (the closeness of independent estimates, their repeatability, and/or the lowest error between estimates) (Barnes, 2001; Morellet et al., 2011) of the tested methods.

5.2. Materials and Methodology

5.2.a. Study sites

In order to evaluate the impact index method for open hill habitats, this study was carried out across three upland areas in northern England: Ilkley Moor, Yorkshire (53.913200, -1.813539); Hole of Horcum, Yorkshire (54.333894, -0.709795); and Borrowdale and Thirlmere, Cumbria (54.552677, -3.106075). Ilkley Moor is an approximately 6.76km^2 open range expanse of dry heather moorland and coarse (acid) grassland. Ilkley Moor is a multiple-use area for conservation action, recreational walking, and livestock grazing, with transient populations of roe deer and resident grazing livestock across the landscape (Friends of Ilkley Moor, 2026). Hole of Horcum is an approximately 2.87km^2 expanse of dry heathland, with some grazing pasture and periphery woodland. This is a multiple-use landscape for farming, anthropogenic activity, and heathland restoration, with a transient population of red deer, and resident populations of roe deer and grazing livestock (Pers Comm, Lawson, J). The Borrowdale and Thirlmere landscape has a 23km^2 open hill expanse of wet and dry heather moorland, and coarse (acid) grassland. There are seven distinct broadleaf deciduous woodlands and three conifer plantation blocks of approximately 17km^2. Borrowdale and Thirlmere is a multiple-use landscape, focussing on nature restoration efforts, water quality management, livestock farming, and anthropogenic activity. The landscape has resident populations of red and roe deer, and grazing livestock.

5.2.b. Open Hill Impact Assessments

5.2.b.1. Index Impact Assessments in Borrowdale and Thirlmere

This work used a plot-based quick assessment index method for open hill habitats attributed to Rory Putman (herein; Putman Method), who adapted and simplified it from the MacDonald et al. (1998a; 1998b) method. Putman Method was adapted to be a quick application method and is designed for use by practitioners (e.g. gamekeepers, farmers, and deer managers) to assess impacts on open hill habitats over the previous 12 months.

Each site surveyed was delineated into ‘areas’ using open access habitats map of the sites, distributed proportionately between the dominant habitats across the sites (e.g. dry/wet heather, coarse grassland). The number of areas surveyed across each site was determined by the availability and proportions of each habitat across the sites. For each area surveyed, 12 randomly distributed 4m^2 stations were surveyed. To maintain this method as a low-complexity survey for practitioners to complete, the 12 stations were selected by throwing a tennis ball within each defined area and the landing point being the centre of the plot. At each station, the surveyor would assign low/moderate/high impact value from a list of candidate impact indicators (Table 5.2) based on the present/available indicators only. Example indicators include consumption of ling heather (*Calluna vulgaris*), blaeberry (*Vaccinium myrtillus*), consumption of bents and fescues, and exposed bare peat from trampling (full list available in Table 5.2). The impact at each station was determined by the modal value, which were averaged across stations to derive the impact score. Modal values were used as the data is ordinal and therefore reduces distortion from outlier results. Low-moderate and moderate-high impacts are assigned when modal values sit between categories.

The age of impacts on open hill habitats were identified *in situ* to increase confidence that recorded impacts had occurred within the past 12 months. This was done by reviewing the growth structure of the plant indicators and comparing with fresh growth for the current growing season. For example, ling heather has a branching growth form at the point of previous consumption, and fresh buds are obvious from April when impact assessments most frequently take place. However, some indicators were assessed purely on their presence due to the time they would naturally persist post-impact (e.g. trampled and damaged sphagnum moss would not likely persist beyond 12 months).

For this study, the Borrowdale and Thirlmere landscape was divided into 12 survey areas: six comprising wet/dry heathland, and six comprising coarse/acid grassland; the landscape was surveyed across three years (April 2022-24). For each area surveyed, the tennis ball selected stations were compared against systematically selected survey stations, using four fixed points equidistant from a plot centre (three plots per survey area) from randomly pre-selected coordinates. Systematically selected station locations were located using a Garmin eTrex 10 GPS unit (Garmin, Kansas, USA; www.garmin.com), with equidistant plots at 30m North, East, South and West from the plot centre, located using a compass and 30m tape measure. All surveys were completed by a single experienced surveyor, who was assisted on Putman Method surveys in 2022 by local deer managers.

Table 5.2. A summary of the impact indicators and descriptors for Putman Method index and intensively sampled impact assessments.

Habitat Type	Indicator	Index Impact Descriptor	Intensive Sampling
Dry and Wet Heathland	**Growth Form:** Individual plants, or heather blanket (ling heather, *Calluna*, only):	• Shaggy-topped [Low] • Smooth cushion/topiary effect [Moderate] • Flat-topped/carpet [High]	Number sub-quadrats with plants : number of sub-quadrats with browsed plants
	Composition: In sites where blaeberry known to be 'potentially present'	• Blaeberry widespread and bushy emergent above heather canopy [Low] • Blaeberry abundant within heather clumps but current year's shoots only [Moderate] • Blaeberry sparse or absent [High]	Number sub-quadrats with blaeberry : number of sub-quadrats with browsed blaeberry
	In sites where bog myrtle present	• Browsed shoots absent/difficult to find [Low] • Browsed shoots easy to find with dense, irregular branching [Moderate] • Closely browsed; dense branching	Number of plants : number of plants browsed
	Direct Impacts: Stem and branch breakage of ling heather	• Not obvious [Low / Moderate] • Conspicuous [High]	Number of sub-quadrats with plants : number of sub-quadrats with broken stems/branches
	Percentage of current year ling heather browsed	• <33% [Low] • 33-66% [Moderate] • >66% [High]	Number of sub-quadrats with plants : number of sub-quadrats browsed
	Erica tetralix consumed	• No [Low] • Yes [High] *(whole plot could be considered high)*	Number of sub-quadrats with plants : number of sub-quadrats browsed
	Erica cineria consumed	• No [Low] • Yes [Moderate / High]	Number of sub-quadrats with plants : number of sub-quadrats browsed

Coarse Grassland	**Composition:** Moss Cover	• Sparse [Low] • <50% cover [Moderate] • >50% cover [High]	Percentage of sub-quadrats with moss
	Bare Ground	• None / very local [Low] • Mostly absent [Moderate/High]	Percentage of sub-quadrats with bare ground
	Direct Impacts: Average sward height	• >6cm [Low] • 3-6cm [Moderate] • <3cm [High]	Average sward height within quadrat
	Percentage of palatable grasses grazed (e.g. bents, fescues, wavy hairgrass, sedges)	• <50% [Low] • >50% [Moderate] • Nearly all [High]	Number of sub-quadrats with grass : number of sub-quadrats grazed
	Percentage of unpalatable grasses grazed (e.g. rushes)	• Absent [Low / Moderate] • Present [High]	Number of sub-quadrats with grass : number of sub-quadrats grazed
	grazing of mat grass (*Nardus stricta*)	• Absent [Low / Moderate] • present [High]	Number of sub-quadrats with mat grass : number of sub-quadrats grazed

5.2.b.2. Intensive Impact Assessments

To test the performance of Putman Method, an intensively sampled equivalent method to produce a percentage of impact was devised. To allow direct comparison, areas delineated following the same procedure as Putman Method. Each area was assessed using 20 equidistant plots, with distances between plots calculated following the procedure from Melville et al. (1983) and located using a Garmin eTrex 10 GPS unit. At each plot, data were sampled within a 4m^2 quadrat (divided into 100 20x20cm sub-quadrats) and independently of Putman Method sampling. The ‘indicators’ used to sample each plot were consistent with those used in the Putman Method (e.g. ling heather, blaeberry, sward height), but instead of using qualitative descriptions of impact, they were recorded as quantitative measurements to capture the percentage of impact. For example, ling heather impacts were recorded as a percentage from the number of sub-quadrats with rooted plants (such as ling heather) and the number of sub-quadrats with impact from ruminant ungulate grazing, while damage to mosses was recorded as the number of sub-quadrats with damaged moss from hoofs (Table 5.2). For each plot, the total percentage of impact was recorded, with the average of the 20 plots for each area used to summarise the deer impact percentages of that area. To differentiate between deer and sheep impacts, this study considered the presence of dung and hoofprints, from which deer and sheep can be differentiated (see Cravino et al., 2024; Ramón-Laca et al., 2014; Sánchez-Rojas et a., 2004; Spitzer et al., 2019). This enables plot level differentiation from impacts, but overall ungulate impact scores across survey areas will be a combination of sheep and deer, which cannot always be confidently separated.

These were completed in Borrowdale and Thirlmere (April 2023), Hole of Horcum (March 2024), and Ilkley Moor (May 2023). For representative cover of the available habitats and size of survey sites, Hole of Horcum was divided into four areas of dry heathland, while Ilkley Moor was divided into two areas of dry heathland and two of coarse/acid grassland. For comparison, Putman Method surveys were performed in the same delineated areas between two-five days after intensive sampling was complete. All surveys were completed by a single experienced surveyor.

5.2.b.3. **Open Hill Impact and Relative Activity**

Red deer relative activity for each defined open hill area in Borrowdale and Thirlmere were calculated from trail camera data (Chapter 4). Relative activity for each area was calculated as the sum of red deer camera trap events, divided by the sum of the number of days the cameras were active, divided by the number of cameras present in that area. The relationship between relative activity and impact was analysed for both the Putman Method index impact assessments, and the intensively sampled impact assessments for 2023.

5.2.b.4. **Data Analyses**

<u>Index Impact Assessments</u>

Index data were organised using the modal value of the indexes to derive whole area impacts. The index data from Putman Method impact assessments were ordinal and therefore did not meet parametric test assumptions. The comparisons between the “tennis ball” and the systematically surveyed station selections were analysed using a Wilcoxon signed rank two-tail test. Further, index data were analysed using Generalised Linear Mixed Model (GLMM) with a Poisson error distribution to evaluate changes to impacts over the three-year study. The predictors used to build the models were the study year the survey took place (fixed effects) and the area being surveyed (random effects), with the model comprised of main effects analysis.

Relationship between index impact assessments and intensive impact assessments

The index data from Putman Method impact assessments were ordinal and therefore did not meet parametric test assumptions. Therefore, the relationship between the index and intensive assessments of open hill impacts were analysed using Spearman's rank correlation.

Relationship between relative activity and impact

The trail camera data and intensively sampled impact assessment were continuous data and met parametric analysis assumptions. Therefore, the relationship was evaluated using Pearson's correlation. The Putman Method index impact assessments were ordinal and did not meet parametric test assumptions. Therefore, the relationship between relative activity from trail cameras and index impact assessments were evaluated using Spearman's rank correlation.

All above analyses were performed in R 4.4.1 (R Core Team, 2024) through RStudio (Posit Team, 2024) using the package MASS (Venables & Ripley, 2002) and LME4 (Bates et al., 2015).

5.2.c. Woodland Impact Assessments

5.2.c.1. Index Activity and Impact Assessments

This work used a transect-based quick assessment index method for woodland habitats used widely across England, the Deer Activity and Impact method (DAI). DAI was originally developed by The Deer Initiative Ltd., citing the work of Cooke (2007; 2009). This method was created for use by landowners, practitioners, forestry regulators, and policymakers to quickly assess the activity and impacts of deer in woodlands.

For each woodland surveyed, a single transect was walked to encompass a representative sample of each woodland and was positioned randomly with regards to established footpaths. For each transect, a tally of activity (Table 5.3) and impact (Table 5.4) indicators were recorded. DAI activity indicators include deer seen, dung seen (enabling species identification), couches, scrapes, wallows, and slots/racks (Table 5.3). DAI impact indicators include bark stripping on trees, browse lines, browsed shrubs, browsed saplings/seedlings/coppiced stools, browsing of bramble, and grazing ground flora (Table 5.4). Numerous DAI indicators were recorded as a sign frequency per km (Table 5.3 and 5.4), therefore the distance of the transect was recorded for accurately interpreting impact scores. Depending on the tally/descriptors recorded for each indicator (Table 5.3 and 5.4), that indicator was assigned a low/moderate/high classification. Low-moderate and moderate-high impacts were assigned when modal index values sit between categories.

As with open hill impacts, the age of candidate indicators was identified *in situ* to ensure activity and impact indicators had occurred within 12 months. Surveys were conducted at the beginning of each spring growing season, using the presence new buds on plants to compare with impacts prior to seasonal growth, and by knowledge of the persistence of indicators. For example, dung persists in the environment for up to twelve months, depending on temperature and precipitation (Delisle et al., 2011; Laing et al., 2003), demonstrating that all dung encountered was deposited within twelve months of surveying. Further, fresh damage to the bark of a tree leaves a moist, brown coloured wood as evidence, whilst historic damage turns dry and grey, thus such indication was used to assist with ageing impact.

In woodlands, many impact indicators evaluated can be ascribed to deer only (e.g. antler impact to trees, wallows, and bark stripping). For impacts that overlap with sheep, differentiate between deer and sheep was performed by assessing the presence of dung and hoofprints (as with section 5.2.b.2). This enables plot level differentiation from impacts, and some certainty over clear deer impacts, however impact scores for each woodland may be a combination of sheep and deer.

A total of 10 woodlands were surveyed across the Borrowdale and Thirlmere landscape, seven of which were deciduous Atlantic rainforest (dominated by English oak (*Quercus robur*), elm (*Ulmus minor*), ash (*Fraxinus excelsior*), and alder (*Alnus glutinosa*)), and three conifer plantations. DAI assessments were performed in April 2022 and 2024, and March-April 2023, with each woodland surveyed from a transect of 1-1.5km, measured using a Garmin eTrex 10 GPS unit (Garmin, Kansas, USA; www.garmin.com). All surveys were completed by a single, experienced surveyor, with assistance from local deer managers for consistent long-term monitoring.

Table 5.3. A summary of the indicators to activity levels of deer from DAI. The tally for index scoring

Activity Indicator	Brief description	Index Scoring
Deer Seen	Record how many of each deer species are seen. For highly gregarious species record group detections, for others record individuals.	None: 0 Low: 1-2 Low-Moderate: 3-4 Moderate: 5-7 Moderate-High: 8-9 High: 10+
Dung	Tally in groups of pellets (group = 6 or more pellets).	None: 0 Low: 1-6 Low-Moderate: 7-14 Moderate: 15-21 Moderate-High: 22-29 High: 30+
Couches	Where deer have lain down, leaving flattened or scraped areas of vegetation (often oval shaped).	None: 0 Low: 1-2 Low-Moderate: 3-4 Moderate: 5-7 Moderate-High: 8-9 High: 10+
Scrapes	Often seasonal in nature, and sometimes possible to identify the species. Scrapes are areas of cleared ground using the legs of the animal.	None: 0 Low: 1-2 Low-Moderate: 3-4 Moderate: 5-7 Moderate-High: 8-9 High: 10+
Wallows	Often seasonal in nature, and usually possible to identify deer species (only Red, Fallow and Sika regularly wallow).	None: 0 Low: 1 Low-Moderate: 2 Moderate: 2 Moderate-High: N/A High: 4+

Racks (Deer Paths) and Slots (individual foot marks)	Individual slots can be useful to identify species and may be the only evidence of deer in low densities. Deer racks are more obvious, long lasting and gives indication to pressure of usage. Species responsible for racks can be determined by: dung, slots, height of browsed vegetation, and height of tunnels (particularly common in Muntjac). Racks are divided into categories of: rarely used, lightly used, frequently used, and heavily used.	None: 0 Low: 1-4 Low-Moderate: 5-9 Moderate: 10-14 Moderate-High: 15-19 High: 20+

Note: The tally for index scoring is calculated as the frequency of signs per km

Table 5.4. A summary of the impact indicators and descriptors for DAI and intensively sampled impact assessments.

Impact Indicator	Brief Description	Index Scoring	Intensive Sampling
Fraying	Often seasonal and localised. Fraying is the rubbing of bark from trees using antlers – larger deer species can move onto thrashing (adding broken branches/stems to the bark removal). Only tally fraying <1-year-old.	None: 0 Low: 1-4 Low-Moderate: 5-9 Moderate: 10-14 Moderate-High: 15-19 High: 20+	Nearest 20 trees to the centre of the plot (within 30m) were surveyed for evidence of fraying.
Bark Stripping	Normally associated with Red, Fallow and Sika. With fresh damage, the width of the teeth marks can differentiate deer from rabbits and squirrels. Count individual stems, or clusters as a single appearance.	None: 0 Low: 1 Low-Moderate: 2 Moderate: 3 Moderate-High: 4 High: 5+	Nearest 20 trees to the centre of the plot (within 30m) were surveyed for evidence of bark stripping by ungulates (*not to be mistaken for squirrel bark stripping*).
Broken Stems	Occurs when deer break stems to browse shoots higher than they could reach. The height of the stem can help identify the species.	None: 0 Low: 1-2 Low-Moderate: 3-4 Moderate: 5-7 Moderate-High: 8-9 High: 10+	Nearest 20 trees to the centre of the plot (within 30m) were surveyed for evidence of broken stems.
Browse Line	May be clearest with leaves on the trees but can be clear in winter. The height of browsing can indicate the species present at the time. Moderate deer densities can lead to the expectation of a browse line visible when looking through the wood at 50-180cm.	Tally and modal scores	Nearest 20 trees to the centre of the plot (within 30m) were surveyed for evidence of identifiable hard pallet mouthed ungulate (e.g. deer and sheep) predation on 35+% of branches.
Browsing	Deer are selective browsers; therefore, it is important to concentrate on significant plants – e.g. climbing ivy and bramble are invariably browsed when deer are present.	Tally and modal scores	Number of sub-quadrats with rooted plants : number of sub quadrats with plants browsed (e.g. ivy) by deer/sheep within a $4m^2$ quadrat.
Coppice <2m (<1m where only muntjac are present)	Recently coppiced stools with all new growth or older growth approximately <2m. They may be individual or in groups. Examine at least 20 representative stools throughout the site, estimating the percentage of stems	Tally and modal scores	Nearest 20 tree stumps/coppiced trees to the centre of the plot (within 30m) were surveyed for evidence of identifiable browsing by hard pallet mouthed ungulates (e.g. deer and sheep).

	with damaged shoots. The more evidence of higher percentage increases the intensity of the impacts.		
Live basal shoots or older coppice or tree boles	Old coppice stools and mature trees continue to produce new shoots from the base, the tops of which are within reach of deer. Look at >20 representative stools/trees spread throughout the site and estimate the percentage of new/live shoots that are damaged. Each stool gets a mark in the percentage ranges.	Tally and modal scores	Nearest 20 tree stumps/coppiced trees to the centre of the plot (within 30m) were surveyed for evidence of identifiable hard pallet mouthed ungulate (e.g. deer and sheep) predation.
Seedling/saplings	Sample no less than 20 seedlings at each area, randomly selected, to stop to look for damage. Each group of 20 is a tally mark in a percentage region. The highest marked area determines a score between Low-High.	Tally and modal scores	Nearest 20 tree seedlings/saplings to the centre of the plot (within 30m) were surveyed for evidence of identifiable hard pallet mouthed ungulate (e.g. deer and sheep) predation.
Bramble	Most common species of bramble are highly palatable to deer so a good indicator of impacts. Between a range of little browsing and most/all browsed.	Tally and modal scores	Number of sub-quadrats with rooted plants : number of sub quadrats with bramble browsed by deer/sheep within a $4m^2$ quadrat.
Grazing	Deer of all species selective graze ground flora. A list of palatable plant species to be used as an indicator.	Tally and modal scores	Number of sub-quadrats with rooted plants : number of sub quadrats with plants grazed (e.g. blaeberry, grasses, and bluebells) by deer/sheep within a $4m^2$ quadrat.

Notes: Index impact scores from tally are calculated as a sign frequency per km. For tally and modal scores, the data collection sheets from the DAI method have specific descriptors to tally to indicate impact level per indicator.

5.2.c.2. **Intensively Sampled Woodland Impact Assessment**

To test the performance of DAI, an intensively sampled equivalent to the impact indicators for DAI to produce a percentage impact was devised. The same woodlands were surveyed as with the DAI method to enable direct comparison in performance. Each area was assessed using 20 equidistant plots within each woodland, with distances between plots calculated following the procedure of Melville et al. (1983) and located using a Garmin eTrex 10 GPS unit. The indicators used to sample each plot were consistent to those use in DAI (e.g. bark stripping, browse lines, browsing, grazing), but instead of tallying qualitative descriptors of impact, they were recorded as quantitative measurements to capture the percentage of impact. At each plot, data were collected for the impacts of browsing and grazing ungulates (predominantly deer) to tree species and ground flora. For tree-based impacts, a nearest neighbour method was used for the 20 nearest trees to the centre of the plot, within 30m of the centre of the plot. Each tree was examined for evidence of bark stripping, fraying/thrashing, browsing of 35+% of branches at deer head height as evidence of browse lines, and browsing of basal shoots. Ground flora was sampled within a $4m^2$ quadrat (divided into 100 20x20cm sub-quadrats) and independently of DAI assessments. The grazing of grasses and both palatable and non-palatable plants was calculated as a percentage of the number of quadrants within which each plant was present and within which it was browsed by deer (Table 5.4). For each plot, the total percentage of deer impact was recorded, with the average of the 20 plots for each woodland used to summarise the deer impact percentage for that woodland.

DAI Activity indicators were compared against an independent sample of relative activity, measured using trail cameras in each woodland (see chapter 4). In each woodland, 2-3 trail cameras (dependent on woodland size) were distributed to encompass a representative sample of each habitat (Figure 4.4). These cameras were active *in situ* for 35-37 days in March-April 2023 to collect data on animal detection events (positive identification with a two-minute shutdown between species identification).

Intensive sampling of woodlands in Borrowdale and Thirlmere took place in March 2023. To enable comparison, the DAI surveys were completed no more than seven days after each woodland had been intensively surveyed. All intensive surveys were completed by a single experienced surveyor.

5.2.c.3. **Woodland Impact and Relative Activity**

The trail camera data enabled independent calculations of relative activity from red deer for each woodland in Borrowdale and Thirlmere. Relative activity for each area was calculated as the sum of red deer camera trap events, divided by the sum of the number of days the cameras were active, divided by the number of cameras present in that area. The relationship between relative activity and impact was analysed by both the DAI index impact assessments, and the intensively sampled impact assessments for 2023.

5.2.c.4. **Data Analyses**

Index Impact Assessments

Index data were organised using the modal value of the indexes to derive whole area impacts. The index data from DAI activity and impact assessments were ordinal and therefore did not meet parametric test assumptions. Index data were analysed using Generalised Linear Mixed Model (GLMM) with Poisson error distribution to evaluate changes to activity and impacts over the three-year study. The predictors used to build the models were the study year the survey took place (fixed effects) and the woodland being surveyed (random effects), with the model comprised of main effects analysis.

Relationship between index impact assessments and intensive impact assessments

DAI data were ordinal and did not meet parametric test assumptions. Therefore, the relationship between DAI Activity assessments to relative activity from trail cameras, and between DAI Impact assessments to intensively sampled impact assessments, were analysed using Spearman's rank correlation.

Relationship between relative activity and impact

The trail camera data and intensively sampled impact assessment were scaled data and met parametric analysis assumptions. Therefore, the relationship was evaluated using Pearson's correlation. The DAI impact assessments were ordinal and did not meet parametric test assumptions. Therefore, the relationship between relative activity from trail cameras and index impact assessments were evaluated using Spearman's rank correlation.

The above analyses were performed in R 4.4.1 (R Core Team, 2024) through RStudio (Posit Team, 2024) using the package MASS (Venables & Ripley, 2002) and LME4 (Bates et al., 2015).

5.3. Results

5.3.a. Open Hill Impact Assessments

5.3.a.1. Index Impact Assessments

From 2022-2024, there were 12 open range areas surveyed in Borrowdale and Thirlmere. Each yeah, there were 144 stations surveyed, 72 selected by tennis ball and 72 by pre-determined GPS locations. The modal impact score for Borrowdale and Thirlmere was moderate in 2022 and 2023, and low-moderate in 2024 (Table 5.5). Using a Wilcoxon signed-rank test, there was no difference in impact score between the use of a tennis ball or systematically selected survey stations ($W = 1.5$, $p = 1$; Figure 5.1). Further, GLMM analysis estimated no significant difference between the average impact levels in Borrowdale and Thirlmere across the three years surveyed ($\chi^2 = -0.497$, $p = 0.619$; Figure 5.1).

Table 5.5. Average open hill impact by area across Borrowdale and Thirlmere.

Area	Modal Impact		
	2022	2023	2024
1	3	3	3
2	2	1	2
3	3	3	3
4	5	5	5
5	5	4	5
6	2	2	2
7	3	3	3
8	3	3	3
9	2	2	2
10	2	3	2
11	2	3	2
12	2	2	2
Site Modal Impacts	3	3	2

Note: Index scores are: 0 = None 1 = Low; 2 = Low-Moderate; 3 = Moderate; 4 = Moderate-High; 5 = High.

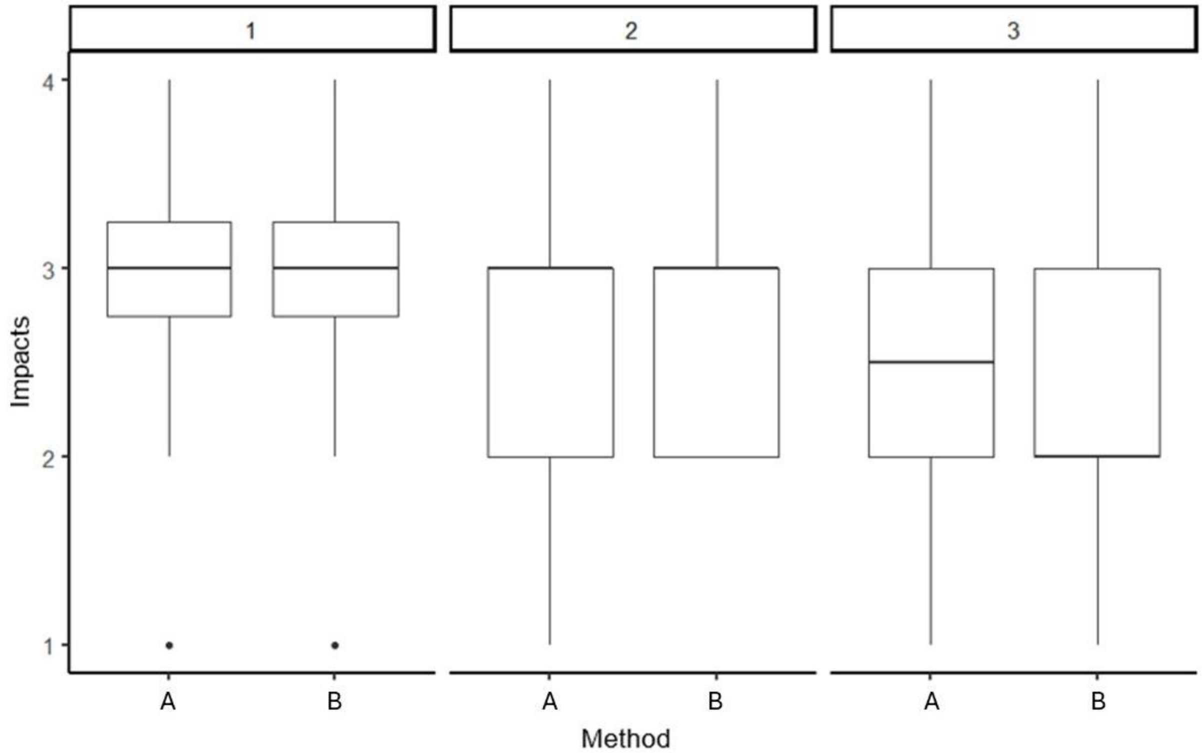

Figure 5.1. Comparison of the compass and tennis ball station selections for index deer impact assessments for the Open Hill habitats in Borrowdale and Thirlmere. Box plots show the median (horizontal line), 75% quartile (box) and 90% quartile (whiskers) of the impacts. Top axis (1, 2, 3) is the year of the study. Y-axis labels are ordinal impact categories: 1 = low; 2 = low-moderate; 3 = moderate; 4 = moderate to high. No high impacts (5) were measured in this study. X-axis is the binary method of station selection; A = Systematically selected stations; B = tennis ball selected stations.

5.3.a.2. <u>Relationship between index and intensive impact assessments</u>

In total, 400 intensive sample plots were surveyed, divided across 20 survey areas between the three landscapes of Borrowdale and Thirlmere (12 areas), Ilkley Moor (4 areas) and Hole of Horcum (4 areas). Putman Method surveys were carried out at Ilkley Moor and Hole of Horcum across the same areas after several days, with a total of 48 stations across the two sites. Intensive sampling data were taken as an average percentage of impact per area, based on available indicators (Table 5.2). Each area took between 6-9 hours to survey, with the signs of impact attributed to a combination of red deer (Borrowdale and Thirlmere, and Hole of Horcum), roe deer (Ilkley Moor and Hole of Horcum), cattle (Borrowdale and Thirlmere, and Hole of Horcum), and sheep (all sites), based on observations of animals and identifiable dung.

There was a strong positive correlation between Putman Method and intensively sampled open hill impact assessments ($r_s(38) = 0.893$, $p < 0.001$; Figure. 5.2). For each habitat evaluated, there was a strong positive correlation for coarse grassland habitats ($r_s(14) = 0.889$, $p = 0.003$) and heather (wet and dry) habitats ($r_s(22) = 0.86$, $p < 0.001$). For individual indicators, bog myrtle was not recorded in enough areas to analyse (n=2), whilst the indicators of ling heather form and browsing, blaeberry, sward heights, grazing grasses, and grazing *Nardus stricta* were more strongly associated to the overall impact scores compared to other indicators (Table E.2).

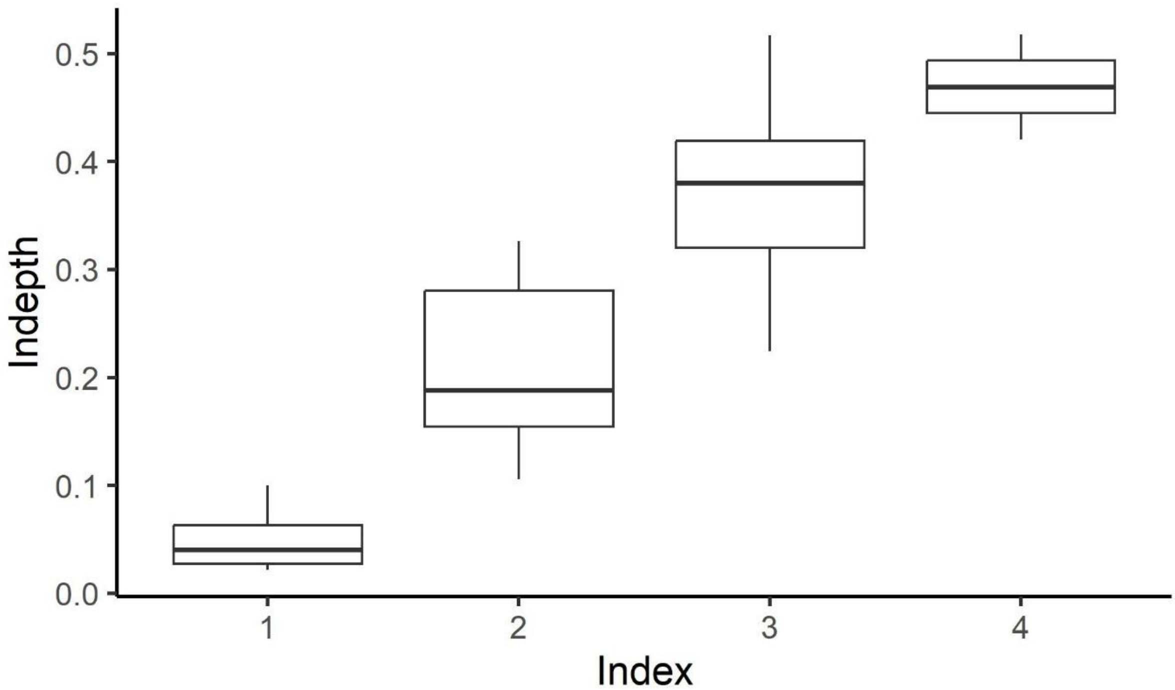

Figure 5.2. Comparison of the Index and In-Depth deer impact assessment methods for Open Hill habitats. Box plots show the median (horizontal line), 75% quartile (box) and 90% quartile (whiskers) of the impacts. X-axis labels are ordinal impact categories: 1 = low; 2 = low-moderate; 3 = moderate; 4 = moderate to high. No high impact areas were measured in this study.

5.3.a.3. Relationship between relative activity and impact

In Borrowdale and Thirlmere, a total of 16 trail cameras (see chapter 4 for distribution) were active in the 12 areas for open hill assessments. In 2023, there was a total of 52 deer detections from trail cameras from 570 camera trapping days. There was no significant relationship between relative activity from trail cameras and the Putman Method index impact assessments ($r_s(10) = 0.199$, $p = 0.536$). Further, there was no significant relationship between activity from trail cameras and the intensive impact assessments ($r(10) = 0.069$, $p = 0.831$).

5.3.b. Woodland Activity and Impact Assessments

5.3.b.1. Index Activity and Impact Assessments

Across Borrowdale and Thirlmere a single transect was surveyed in each of 10 woodlands in 2022 and 2023, and seven woodlands in 2024 (due to access restrictions). The modal activity score for the site was low-moderate in 2022, and low in 2023 and 2024 (Table 5.6). The modal impact score for the site was low-moderate in 2022 and 2023, and moderate in 2024 (Table 5.6). There was no significant trend in DAI scores over the three years (Table E2) for Activity ($\chi^2 = -0.699$, $p = 0.485$) or Impact ($\chi^2 = 0.346$, $p = 0.729$).

Table 5.6. Average DAI woodland activity and impact scores by area across Borrowdale and Thirlmere.

Woodland	2022		2023		2024	
	Activity	Impact	Activity	Impact	Activity	Impact
1	1	1	1	1	1	1
2	2	2	1	1	1	1
3	4	4	3	3	1	3
4	5	4	5	5	3	3
5	1	2	1	1	1	3
6	1	2	1	3	2	3
7	2	2	2	3	3	3
8	1	1	1	1	N/A	N/A
9	1	1	1	1	N/A	N/A
10	4	2	4	2	N/A	N/A
Side Modal Score	2	2	1	2	1	3

Note: Index scores are: 0 = None 1 = Low; 2 = Low-Moderate; 3 = Moderate; 4 = Moderate-High; 5 = High.

5.3.b.2. Relationship between index and intensive activity and impact assessments

For intensive sampling, 200 plots were surveyed across the 10 woodlands in Borrowdale and Thirlmere. Intensive sampling data were taken as percentages of impact per area, with a mean impact of 0.166 (sd = 0.097) for the site. Each woodland took between 10-14 hours to survey, with the signs of impact attributed to a combination of red deer, roe deer, and sheep, based on observations of animals, identifiable dung in each woodland.

There was no relationship found between the DAI activity index and trail camera estimates of relative activity ($r_s(18) = 0.23$, $p = 0.53$). There was a significant strong positive relationship between DAI impact assessments and intensively sampled assessments of woodland impact ($r_s(18) = 0.809$, $p < 0.05$; Figure. 5.3). For the impact indicators, bark stripping was not recorded in enough woodlands to analyse (n=2), whilst impacts to coppice/basal/saplings, bramble, and grazing were more strongly associated with the overall impact scores compared to other indicators (Table E.2).

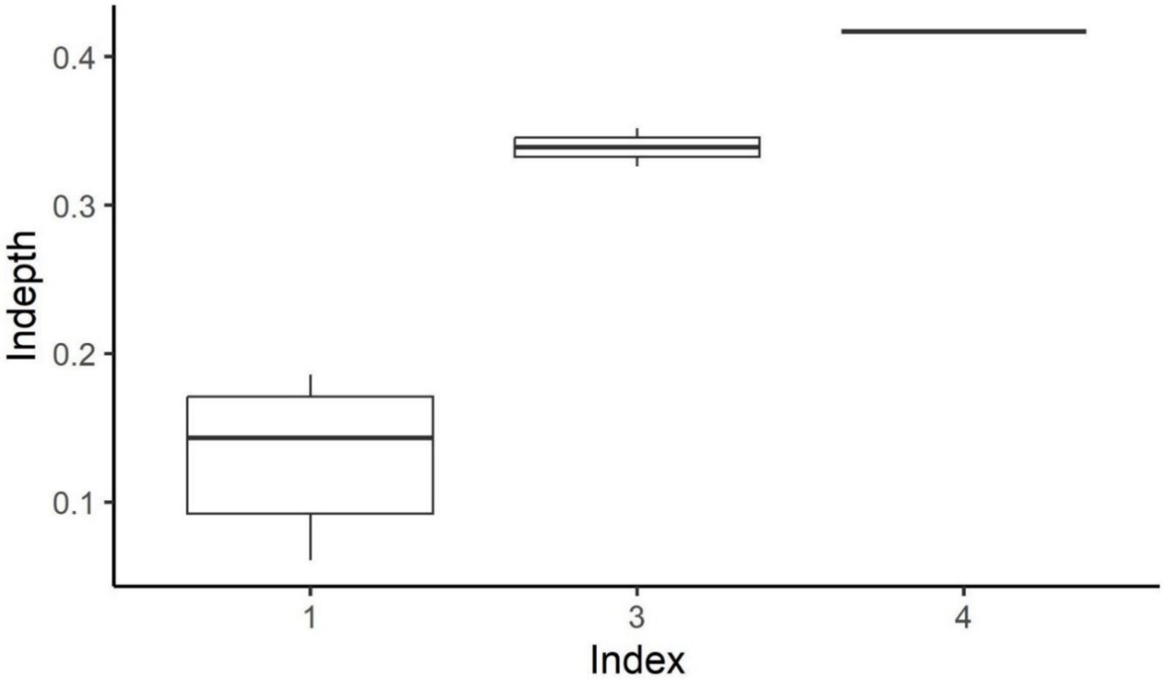

Figure 5.3. Comparison of DAI impact and intensively sampled deer impact assessment methods for 10 woodlands. Box plots show the median (horizontal line), 75% quartile (box), and 90% quartile (whiskers) of the impacts. X-axis labels are ordinal impact categories: 1 = low; 3 = moderate; 4 = moderate to high. No low-moderate (2) or high (5) impacts were measured in this study.

5.3.b.3. Relationship between relative activity and impact

In Borrowdale and Thirlmere, a total of 21 trail cameras (see chapter 4 for distribution) were active in the 10 woodlands assessed for impacts. in 2023, there was a total of 120 deer detections from trails cameras across 757 camera trapping days. There was no significant relationship between relative activity from trail cameras and DAI index impact assessments ($r_s(8) = -0.023$, $p = 0.951$). Further, there was no significant relationship between activity from trail cameras and the intensive woodland impact assessments ($r(8) = 0.243$, $p = 0.499$).

5.4. Discussion

The hypothesis that a well performing indexed impact method would have a strong positive and significant correlation to intensively sampled equivalents was supported for the impact indicators for DAI and the Putman Method. Moriarty et al. (2018) determined that index and scale-measured indicators significantly correlating at $r \geq 0.7+$ would constitute an effective and reliable index method; DAI and Putman Method exceeded that. We further identified that the influential indicators for DAI and intensive assessments were impacts to coppice/basal/saplings, bramble, and grazing (Table E.1). The influential indicators to the Putman Method were ling heather growth form and browsing, sward heights, grazing, and *Nardus stricta* (Table E.1). Moderately correlating indicators such as blaeberry (Table E.1) were inconsistently impacted between low-moderate and moderate-high. Blaeberry is a high value food source to deer (see Spitzer et al., 2021) and has been noted to be disproportionately targeted for consumption by deer compared to the availability of other food sources (Bobrowski et al., 2020). Therefore, the relationship between blaeberry (and other moderately correlating impact indicators) to impact and plot/area impact may be non-linear, where they may be highly impacted in areas that are otherwise lowly impacted, as a result of preferential feeding as a higher value food source.

Across the three years of study there were no significant changes to activity and impact (woodlands) or open hill impact indexes. It is possible that no changes occurred over the three years, or that the methods are not sensitive enough to measure any small changes that might have occurred. The coefficient of variation for the index indicators for DAI Impact (0.514), and Putman Method impact (0.325) show low precision for estimates and therefore should be further evaluated to the ability of the indexes to detect fine-scale annual changes to impact in the environment. It is also likely to be reflective of the complexities and temporal-scale, which can exceed 10 years (Tanentzap et al., 2011), for floral species and compositions to recover from herbivore browsing/grazing. Complexities obstructing recovery includes landscape heterogeneity, soil nutrition, small mammal population recovery increasing seed predation, and seed propagation for low populations of preferred browse species (see Royo et al., 2010; Tanentzap et al., 2011; 2012; Reed et al., 2022). Thus, simply removing or reducing deer populations would not necessarily return an area to an unimpacted state (Coomes et al., 2003).

A key component of the Putman Method was the selection of random stations within a stratified area with a tennis balls, however there is an implied risk of bias from deliberate placement. The effect of site-selection bias is well described (e.g. Mentges et al., 2021), hence our decision to evaluate the tennis ball approach. Putman Method areas were stratified to dominant habitat cover, and multiple plots were surveyed to obtain a modal value of impact. Results in this work has demonstrated that, without deliberate placement, the tennis ball can select a representative sample of the survey area and reduce site selection biases described by Mentges et al. (2021). We found this through no significant difference to index impact scores using the Putman Method between the pre-determined surveyed stations and tennis ball selected stations (Figure 5.1). There was some variation between the tennis ball and pre-determined plots in years 2 and 3, including different median impact values in year 3 (Figure 5.1), but not a significant difference. The tennis ball was not perfectly consistent, but the differences were not significant. Thus the tennis ball can be expected to deliver impact assessments that are consistent with a more intensive structured sampling method, and since it requires less effort, may be preferable to practitioners.

The Borrowdale and Thirlmere red deer population mean abundance had decreased by approximately 5.7% per year over the 3 years studied (Chapter 4), with no known changes to local livestock populations. Previous studies have demonstrated a relationship between deer density and impact across hundreds to thousands of km^2 (Albon et al., 2007), but not on sites less than hundreds of km^2, and weak/no relationship between density and grazing impacts (Albon et al., 2007; Blossey et al., 2019; Moore et al., 2018; Pierson & deCalesta, 2015). Instead, characteristics of the environment and habitat structures have significantly more effect on impact scores over population density (see Moore et al., 2018). This is also linked to the size of continuously connected areas, which has a direct impact upon how resilient habitats are to impact (Albon et al., 2007). Further, Moore et al. (2018) identified livestock grazing as one of several anthropogenic pressures incorporated into habitat impact assessments, with its contribution to overall impact scores depending on local context and landscape management intensity. As such, with a reduction to the deer population abundance, and no change to the livestock population, a reduction of impact in Borrowdale and Thirlmere over three years would not be directly expected.

Based on the strength of the relationships between index and intensively sampled impact assessments, the lack of a reduction to impact was therefore likely a product of complexities to floral species recovery lags (described in MacDonald et al., 1998a; Tanentzap et al., 2012) and other environmental factors (Moore et al., 2018), rather than the performance of the index methods. These factors include livestock grazing upon the open hill and some woodland habitats, consuming overlapping resources to deer (Fraser et al., 2009). Moore et al. (2018) identified that open range habitat impact assessment methods (not including the Putman Method) were effective in identifying mosaics of impacts across landscapes, including the difference between areas with and without wild herbivores and livestock. However, as there was only one open hill area without sheep present, and no data available on sheep stocking density, differences could not be concluded in this work. Instead, the combined impacts of deer and sheep were quantified, and future work should be planned to evaluate whether the Putman Method captures habitat mosaics as described by Moore et al. (2018) and can discriminate effectively between deer and sheep impacts.

Of the open range habitat impact assessment methods available in GB, to date none have been independently validated as methods for their accuracy or precision. Of the available habitat impact assessment methods used in Scotland (not including the Putman Method), Moore et al. (2018) concluded that the methods described impacts well on a spatial scale, but commonly overemphasised damage. Consequently, the existing methods for open range impact assessment evaluated by Moore et al. (2018) may currently be overstating the extent of habitat degradation, reducing their reliability for evaluating habitat conditions. Further, the existing methods have not been subjected to independent evaluation, meaning that their accuracy and precision are unverified. In contrast, the Putman Method correlated strongly with our independent assessment of impact, meeting the validation threshold set out by Moriarty et al. (2018) to be considered an independently validated index method. The Putman Method is also low-complexity to undertake and analyse, which is ideal for practitioner uptake, and is considerably faster to complete (1-3 hours for Putman Method, 6-9 hours for intensive), with simpler data collection and analyses than our independent impact assessment method. These attributes suggest that the Putman Method may currently be well-suited to practitioner needs (see Maffey et al., 2016; Månsson et al., 2023; Palmer et al., 2021) and improve consistency in approaches to measuring open range impact.

In woodland habitats, the independent measurement of Activity using trail cameras did not correlate with the DAI Activity index used in woodlands, however it would be premature to conclude that it therefore is not an acceptable broad indicator. The independent assessment of activity was derived from trail camera deployments. Relative activity was then calculated from 35-37 days (per camera) of camera deployments, whilst the index indicators are based on up to or at least 12-months of accumulated signs of activity (e.g. dung accumulates over months, browse lines and wallows develop over years). Red and roe deer have different seasonal activity patterns (Pépin et al., 2009; Pagon et al., 2013); with red deer being more active in woodlands during the winter compared to spring and summer (Pépin et al., 2009). Thus, estimating relative activity across 6 weeks in spring would likely miss annual activity variation. As the independent woodland assessment of activity did not directly compare against each activity indicator, unlike impact validations, it does not meet the recommendations for validating index methods set out by Moriarty et al. (2018).

From this work (Chapter 4), an assessment of relative activity from trail cameras demonstrated that the red deer population were not highly active in most woodlands at the time of their deployment. As activity indicators by DAI demonstrate accumulations of activity, they may not be reflective of the current activity levels from deer at the time of surveying. Further, in both woodland and open range environments, there was no significant correlation between relative activity from trail cameras and impact (either index or intensive). These findings were consistent with other studies that demonstrated a non-linear relationship between relative activity and impact (e.g. Blossey et al., 2019; Pierson & deCalesta, 2015; Wójcicki et al., 2023). Therefore, although the activity indicators were not validated for DAI, their use as a tool to assist with DAI surveys is questionable. The clearest benefit that could be recommended would only be as a presence/absence indicator of species rather than as a reportable index figure.

Multiple attempts have previously been made to create index methods for assessing deer impacts in woodland habitats, including DAI transects (Forestry Commission, 2023), and multiple plot-based indexes (e.g. Injima & Nagaike, 2015; Pierson and deCalesta, 2015; Wild Deer Best Practice Guide for Scotland, 2018; Armstrong, 2023; NatureScot, 2024; Table 5.1). Until now, none of the above methods have had a validation assessment evaluating method performance. The approach of Melville et al. (1983) intensively surveyed a representative sample of the landscape in forestry units, but required significant resource investment to complete, and the approach has not been taken up by practitioners. Further, the method advocated by NatureScot (2024) requires annual returns to the same plots, which is resource intensive. Conversely, DAI appears to suit practitioner needs through low resource investment with acceptable findings (see Maffey et al., 2016; Månsson et al., 2023; Palmer et al., 2021), although DAI risks not surveying a representative sample across each woodland from biased transect placement (see Mentges et al., 2021). A transect-based method is likely to be considered a simpler, less complex, and user-friendly data collection approach for assessing deer impacts when compared to systematic, annual return plots. As such, the DAI method is more likely to be undertaken in the field by practitioners over less simple methods. What remains unclear is whether DAI would lend itself to surveying similar woodland habitats, with localised adjustments to indicators, in other regions (e.g. continental Europe, East/Southeast Asia, Australia and New Zealand, and North America).

We have demonstrated, through validation, that DAI is likely a reliable method for confidently estimating impacts to woodland habitats and can be completed in approximately 2-4 hours per DAI transect, which is considerably faster than the approximately 10-14 hours for each intensively sampled woodland. As this work only evaluated 10 woodlands, questions linger as to whether the sample size may have influenced results (see Jennions et al., 2002). Further, there were no recorded low-moderate or high impacted sites for the comparison of methods. Without multiple repeats of woodlands at each index value between low and high, comparisons may not have reflected the variation within the data (see Jennions et al., 2002). However, the strong and significant correlations between the intensive and index samples (Table 5.3) suggests acceptable accuracy and precision. As such, these findings support the use of DAI for assessing woodland impacts (see Benedetti-Cecchi, 2003) suggest it would be reliable for practitioners to uptake and use of the method (see Maffey et al., 2016; Månsson et al., 2023; Palmer et al., 2021). Remaining areas of research for DAI include assessing the performance of the method in areas of higher and of lower impact levels, and temporal monitoring numerous areas to evaluate whether the lack of change over time was due to no change or a lack of sensitivity in the indexes.

From both the DAI and Putman Method, the key metrics measured are impacts from herbivorous ungulates at multiple scales, including the impact to specific indicators, the average impact to areas/woodlands, and the average impact across a site. The methods are a combination of deer specific indicators (such as antler impact to trees), and more generic ungulate impact (such as grazing). The key method for differentiating between generic ungulate impacts is the presence of dung and hoofprints, from which deer and sheep can be differentiated (Cravino et al., 2024; Ramón-Laca et al., 2014; Sánchez-Rojas et a., 2004; Spitzer et al., 2019). However, drawing conclusions only from dung and hoofprints is cautioned against due to inconsistencies and observer bias (Cravino et al., 2024; Ramón-Laca et al., 2014;

Spitzer et al., 2019), therefore clearer identifiers of the causes of impact need to be evaluated through experimental assessments of grazing patterns, behaviours, and signs.

Habitat impact assessments, such as DAI and Putman Method, are designed to evaluate whether deer populations (and other ecologically similar animals, including livestock) are ecologically sustainable in the context of anthropogenic landscape management objectives. These methods typically seek to capture realised ecological effects, particularly impacts on habitat regeneration, plant community composition, and structural diversity. Their importance is most evident within conservation-sensitive habitats and as part of wider landscape-scale management frameworks (Champagne et al., 2021; Reimosser & Nopp-Mayr., 2024). As index-based approaches, habitat impact assessments are effective at identifying cumulative effects over time and at signalling whether current grazing or browsing pressure is incompatible with desired habitat outcomes. However, they are subject to several recognised limitations, including observed impacts reflecting legacy pressures due to ecological time-lags, and the methods are not currently capable of reliably attributing impacts to specific herbivore species where multiple grazers coexist (i.e. unreliability from using dung and hoofprints; e.g. Cravino et al., 2024; Ramón-Laca et al., 2014; Spitzer et al., 2019). Despite these limitations, habitat impact assessments remain valuable tools for determining whether grazing and browsing pressures are constraining wider landscape objectives, and for informing target-setting for wildlife management targets (Champagne et al., 2021; Reimosser & Nopp-Mayr., 2024).

At present, many habitat impact assessment implicitly treat habitat vulnerability as uniform, despite clear evidence that resilience to herbivory varies with environmental conditions, habitat patch size, species richness, and vegetation structure (Moore et al., 2018). Methodological improvements could therefore include more explicit consideration of habitat extent and context (partly address by DAI through sign frequencies per km) and recorded historic impacts, and clear age-structured indicators to determine regeneration. Where reductions in impact scores achieved through deer management do not translate into desired habitat recovery, additional habitat-specific management interventions are likely to be required alongside continued herbivore management (Reimosser & Nopp-Mayr., 2024). These may include clearly defined livestock grazing controls and active habitat restoration measures (e.g. replanting or other techniques). Accordingly, validated index habitat impact assessments, such as DAI and Putman Method, should function as decision-support tools within integrated landscape management frameworks, informing deer management alongside, rather than in place of, other ecological restoration and land-use management measures.

5.5. Conclusion

Indexed habitat impact assessment methods can provide reliable estimates of ecological impacts of herbivory at landscape scales when independently validated against intensive sampling. Both the DAI in woodland habitats and the Putman Method in open hill habitats had strong, significant correlations with intensively sampled impact assessments, exceeding established validation thresholds and confirming their effectiveness as indicators of habitat condition. The results further indicated that activity or abundance were weak and non-linear predictors of impact in heterogeneous, multi-herbivore landscapes. The absence of measurable reductions in habitat impact despite declining deer abundance over the study period highlighted the importance of ecological recovery lags, environmental constraints, and overlapping pressures from livestock grazing, demonstrating that changes in deer numbers alone should not be expected to produce short-term habitat recovery. Habitat impact assessments function most appropriately as measures of ecological sustainability relative to landscape management objectives, rather than as tools for establishing direct causal links between individual herbivore species and observed impacts.

Crucially, indexed methods such as DAI and the Putman Method are particularly well suited to applied land-management contexts due to their low complexity, transparency, and efficiency. Compared with intensive survey approaches, these methods require substantially less field time, simpler data collection, and minimal analytical processing, while still producing ecologically meaningful results. The validation of the tennis-ball plot selection approach to Putman Method further demonstrates that simple, randomised sampling designs can reduce site-selection bias without increasing survey burden, enhancing consistency and repeatability for practitioners. These characteristics align strongly with practitioner needs, increasing the likelihood of routine implementation and long-term monitoring across large or remote landscapes. However, the study also identified important limitations, notably the implicit treatment of habitat vulnerability as uniform despite clear variation in resilience driven by habitat size, structure, productivity, and legacy condition. Incorporating habitat context, historic impacts, and age-structured regeneration indicators would improve the ecological sensitivity of these frameworks. Ultimately, indexed habitat impact assessments should be understood as user-friendly decision-support tools within integrated landscape management, informing deer and livestock management decisions alongside (rather than in place of) broader habitat restoration and land-use interventions necessary to achieve long-term ecological recovery.

Chapter 6

An interdisciplinary approach to achieve consensus for evidence-based management of red deer in the Lake District

6.1. Introduction

Human-wildlife conflicts are considered a global conservation concern by the UN Convention on Biological Diversity post-2020 (IUCN, 2022). The human-human conflicts inherent within human-wildlife conflicts are gaining increasing research interest as understanding conflict between human groups is essential if solutions are to be found to human-wildlife conflicts. Yet, many studies continue to address these conflicts from either a natural science or social science approach, rather than an interdisciplinary approach to identify and evaluate the conflict holistically (advocated for by Zimmerman & Stevens 2021). This PhD applied an interdisciplinary approach to assess the biological condition and ecological impacts of the red deer (*Cervus elaphus*) population of the Borrowdale and Thirlmere landscape. Further, this PhD also developed an exemplar of best practice for achieving consensus in collaborative, landscape-scale management by integrating stakeholder values with facilitated shared decision-making. This thesis has demonstrated a robust framework for identifying and defining both human-wildlife and human-human conflicts, supports the co-creation of conflict mitigation strategies, and provides methodological recommendations that balance high-quality data collection with usability to support long-term monitoring.

Red deer are a charismatic UK native species (Mattioli et al., 2022) that are valued for the emotional and physical well-being their presence provides people (Chapter 2), as well as their ecological and economic value (Nilsen et al., 2007; Pérez-Espona et al., 2013; Putman, 2010; 2024). Across Great Britain, red deer have substantially expanded in range since the 1970's (Ward, 2005; Figure 1.3), with increased reporting of deer impacts across their range (e.g. NatureScot, 2024). This expanded range and increased deer impact risk directly conflicts with the UK government's targets of woodland creation for carbon capture and biodiversity benefits (DEFRA, 2018).

Therefore, to improve the likelihood of achieving UK government targets for nature restoration (e.g. DEFRA, 2018), the management of browsing herbivores that impact woodland creation, such as native red deer, is essential. The areas identified for woodland creation efforts are spread across UK rural landscapes that overlap red deer populations, thus demonstrating the risk of deer impacts to accomplishing government nature restoration targets.

6.2. Values and decisions from a single facilitated process

Human-wildlife conflicts (HWCs) occur where people are unable reconcile the impacts of wildlife to anthropogenic land-use with human-nature coexistence (IUCN, 2020). Additionally, HWCs are commonly the conflicts between humans about wildlife populations (e.g. Goyes, 2022; Ramos, 2022). To produce lasting mitigations and/or solutions to HWCs, the underlying human-human conflict must be resolved (Goyes, 2022; Ramos, 2022). The variety of ways by which people value nature and wildlife impact how individuals make decisions on conservation issues (e.g. Butler, 2016; Raymond, et al., 2016; Wardropper, et al., 2024). Thus, understanding the motivations, perceptions and values of people, combined with effective communication among stakeholders, is required to establish equitable discussion on contentious issues (Bhatia et al., 2020; Dickman, 2010; Kansky et al., 2016; Madden & McQuinn, 2014; Martínez-Jauregui et al., 2023; Woolaston et al., 2021). Establishing equitable discussion through understanding individual motivation is therefore essential in creating consensus-based, collaboratively produced, mitigations, and for ensuring their support and uptake (Pimid et al., 2022; Salvatori et al., 2020; 2021; Vatn et al., 2024).

Previous work has evaluated the values of people towards conservation problems with the intention of being later used to equitably inform conservation policy and decision-making (e.g. Bieling et al., 2020; Brunetta & Voghera, 2008; Chan et al., 2016; Heindorf et al., 2024; Lees et al., 2023; Raymond et al., 2016; Riechers et al., 2020). Other works have engaged participants to evaluate how stakeholders understand and perceive conservation problems, where conflicts emerged, and how acceptable proposed solutions might be (e.g. Loker et al., 1999; Kilpatrick & LaBonte, 2003; Siemer et al., 2004; Dandy et al., 2009; 2011). Each of these approaches demonstrated one or some elements of those recommended in the previous paragraph, but none demonstrated a fully holistic approach (advocated by Zimmerman & Stevens, 2021) to encompass all elements of the identified conflict. By evaluating only elements of HWCs, creating effective mitigations to the whole conflict will likely be impossible. However, engaging stakeholders involved in a HWC through a bottom-up process (from defining the problem to co-creating mitigation strategies) offers the strongest chance of successful outcomes, yet evidence of this approach being applied elsewhere has not been identified.

The focus of Chapters 2 and 3 of this thesis was to apply a bottom up-approach. The Delphi process used three rounds of questionnaires that facilitated a broad range of stakeholders, as participants, to elicit a wide range of expression using open questions. The use of open questions enabled equitable discussion for decision-making and for evaluating the values and perceptions of participants that influenced decision-making. Chapter 3 demonstrated that engaging stakeholders through a facilitated bottom-up approach can help create definitions and enable discussion for conflicts that culminates in a consensus achieving co-created conflict-mitigation decision-making framework. Importantly Chapter 2 demonstrated that participants did not need to be fully aligned by values and perceptions on all problems to reach agreement through open discussion. Thus, facilitated discussion methods such as Delphi are capable of achieving consensus to mitigate HWCs by reducing human-human conflicts between stakeholders and ensuring their participation throughout the entire the decision-making process.

The Delphi process resulted in consensus among participants on a high-level vision for adaptive deer management, three core objectives underpinning this vision, the necessity of integrating lethal control with exclusion strategies, and the importance of ongoing monitoring of deer population abundance, population dynamics, and ecological impacts across the landscape. A central commitment from the workshop was to incorporate this co-developed framework into a landscape-scale deer management policy for Borrowdale and Thirlmere, embedding key principles of adaptive management (see Williams et al., 2011), open communication with stakeholders, and collaborative decision-making. However, while participants agreed in principle to undertake annual drone surveys and deer impact assessments using index-based methods, there was no clear agreement on responsibility for the planning, funding, or implementation of these monitoring activities. In the absence of a clearly defined ownership of these actions, there is a risk that the agreements reached, following several months of participation and effort across four rounds of Delphi, not being placed into action. may not translate into practice. Addressing this implementation gap will be the focus of a follow-up study, likely conducted through interviews with workshop participants, to evaluate whether and how the agreed framework has influenced policy, identify barriers to implementation, and explore potential means of overcoming these obstacles.

Although some stakeholder groups identified in the participant maps (Tables 2.1 and 3.1) were more represented than others within the Delphi process (e.g. only one tenant farmer), participants were engaged as individuals representing themselves or their organisations rather than as formal representatives of stakeholder categories. Accordingly, the consensus developed reflects the views of the participating actors, informed by iterative questionnaire rounds of facilitated discussion, rather than an attempt to achieve proportional stakeholder representation. The process provided opportunities for participants without formal decision-making authority to contribute directly to discussions involving actors with legal responsibility, while the inclusion of civil servants and contract deer managers (actors not typically involved in strategic decision-making in this area) introduced operational perspectives and broader public objectives. These contributions also reflected perspectives raised by participants in earlier questionnaire rounds who were not present at the final workshop, such that their views remained incorporated into the deliberations despite their absence.

As a method, the Delphi technique may help to bypass pre-existing biases between people to resolve human-human conflicts. This is consistent with the recommendations of van de Water et al. (2023), who found conflict resolution in conservation required a shared vision, collective benefits and relevant (not identical) values. Previous works in the UK have evaluated the social tolerances towards deer culling, yet consensus was never reached, nor solutions created from that research (e.g. Dandy et al., 2011; Kilpatrick & LaBonte, 2004; Siemer et al., 2004). The findings of Chapter 2 and 3 suggest that facilitating discussion between stakeholders about human-wildlife conflicts (e.g. red deer impact management) is essential to demonstrate areas of agreement enabling co-created visions. Had previous works (e.g. Dandy et al., 2011; Kilpatrick & LaBonte, 2004; Siemer et al., 2004) taken an approach similar to Chapter 2 and 3 of this PhD, there may have been the focus and increased opportunity required for consensus to be reached, and possible solutions and/or mitigations developed. Therefore, through strategically designing a Delphi process to the specifics of a conflict, including the stakeholders involved and the technology available, would likely be applicable globally to any landscape-scale HWC.

6.3. Combining and evaluating population estimates and deer impacts for deer management actions

The adaptive impact management framework developed by the participants of this study requires the regular evaluation of both deer impacts and population abundance to inform deer management actions. Following the recommendations of Williams (2011a), adaptive impact management frameworks require stakeholder collaboration, landscape-scale objectives focussed on reducing impact from target species, agreed methods of management, modelling the likely outcomes from management actions, and continuous evaluation of each element. Index deer impact assessments (Chapter 5) would be used to determine whether deer management actions are meeting two of the agreed management objectives (Figure 3.2). Population abundance estimates (Chapter 4) would also be used to inform management actions to meet the objectives set out in Chapter 3 (Figure 3.2). By integrating the evaluation of methods used to estimate deer density (Chapter 4) with those used to assess ecological impacts (Chapter 5), it becomes possible to critically assess whether the monitoring approaches identified by participants in Chapter 3 adequately meet their stated information needs, and to evaluate whether these methods represent the most effective options available. Through identifying monitoring methods that meet stakeholder requirements, the participants may gain increased confidence in the data used achieve adaptive deer impact management objectives across Borrowdale and Thirlmere.

The adaptive management framework co-created in Chapter 3 (Figure 3.2) demonstrated all of the elements recommended by Williams (2011a), apart from population modelling. Thus an area of future research should include evaluating whether different population models provide realistic projections of deer population change. However, such models should be user friendly for non-ecologist practitioners to be able to complete (consistent with Maffey et al., 2016; Månsson et al., 2023; Palmer et al., 2021). Should a model be identified that provides realistic projections of deer population change based on management actions, and is simplistic in use, that model would be recommended to the stakeholders from Chapter 3 to use in their evaluations of the potential impact to population abundance from culling efforts.

The results of Chapter 4 demonstrated, from estimates using REM, that the average density of red deer in Borrowdale and Thirlmere decreased over the three-year period studied by approximately 5.7% per year. Across the three-year study period, the average index impact scores in both woodland and open range habitats did not significantly change. There are numerous reasons why changes to impact score may not have been identified through this time. The first is that the density of deer populations only weakly correlates with deer habitat impact assessments (Moore et al., 2018), thus a reduction of impact over the three years could not have been predicted based on the reduction of deer alone. Secondly, the time between deer impacts occurring and plant species recovering differs between plant species, such as seedlings growing or new branches forming (Tanentzap et al., 2011). Therefore, heavily impacted areas may provide an illusion of high impact for several year after high levels of impact ceased as a result of a lag in recovery time for plants. Because both DAI and the Putman Method assign vegetation impacts to broad categorical states, they are insensitive to gradual or early-stage plant recovery, and may therefore fail to detect recovery processes until sufficient change has occurred to shift an index category, creating an apparent lag in recovery time.

Sheep grazing is known to impact vegetation in ways that are comparable, but not identical, to deer (Clutton-Brock & Albon, 1989; DeGabriel et al., 2011; Hester et al., 2001; Oom et al., 2008). The landscape of interest in this PhD, Borrowdale and Thirlmere, has substantial sheep grazing on open range habitats, with regular encroachment on woodland habitats. There are regular encroachments of sheep to areas they are supposed to be excluded from, likely due to stock exclusion fences being broken, or gates on fences being left open by hill walkers. Not being able to discriminate between the impacts of deer populations and sheep grazing makes it hard to assess whether sheep or deer impact the landscape equally or otherwise. Yet, the presence of sheep causes red deer to modify their spatial behaviour, with deer avoiding areas grazed by sheep (Clutton-Brock & Albon, 1989; DeGabriel et al., 2011), leading to a lack of fixed grazing patterns in upland landscapes (Hester et al., 2001). Therefore, the actions and presence of dense sheep populations grazing on small habitat patches may reduce the localised impacts of deer (Hester et al., 2001), but not the herbivore impact score due to sheep feeding behaviours (Oom et al., 2008). As such, methods of assessing impact and herbivore density/activity need be calibrated to discriminate between sheep and deer impacts effectively. Should there be a further reduction of deer populations across the study landscape, the continued monitoring of ungulate impacts and an increased monitoring of sheep activity may provide clearer indications as to the source of impact. As such, landscape management objectives for this area (aiming for habitat regeneration) must, therefore, include management objectives for both sheep and deer populations. Identifying the relative impact of deer and sheep in the landscape will assist in targeting strategic deer impact management, and wider landscape management to meet landscape objectives and vision for Borrowdale and Thirlmere (Figure 3.2).

6.4. Interdisciplinary approaches to mitigating human-wildlife conflicts

There are increasing calls for interdisciplinary approaches towards management and mitigations of human-wildlife conflicts (e.g. König et al., 2020; Salerno et al., 2025; White & Ward, 2010; Yeshey et al., 2024; Zimmerman & Stevens, 2021). However, it is not clear within existing literature whether these interdisciplinary approaches are consistently put into practice. In the absence of documented interdisciplinary approaches, it is hard for practitioners and scientists to review processes, identify best practices, or evaluate how such approaches could be applied to other conflicts. This PhD demonstrated that interdisciplinary methods could achieve consensus in a co-created adaptive management framework, towards mitigating HWCs, by combining bottom-up stakeholder engagement with ecological evaluations to ensure informed decision-making.

As described in Section 6.2, engaging participants in this PhD (Chapters 2 and 3) facilitated consensus in decision-making in a way that had not previously been achieved in deer management. The natural science elements of this research (Chapters 4 and 5) enabled the evaluation, and subsequent recommendations (see section 6.5) of methods for estimating population size and deer impact assessments. The methods were selected to meet the decisions made by, and needs of, the study's participants (Figure 3.2). Population abundance estimates and impact assessment methods were selected to be user friendly while providing "high quality data" to monitor the population size and habitat impacts of deer across the landscape.

Without using the process of engaging stakeholders as in this thesis, the "information needs" (Chapter 3) of people could not be known and would therefore have been left for academics or practitioners to decide for themselves. Without involving stakeholders in the decision-making process, resolving/mitigating the human-human conflict associated with this HWC would not have been achievable (König et al., 2020; Salerno et al., 2025; White & Ward, 2010; Yeshey et al., 2024; Zimmerman & Stevens, 2021). Importantly, this work demonstrated that mitigating the human-human conflict through bottom-up engagement is achievable (Zimmerman & Stevens, 2021), and from this, co-created mitigations to a HWC can achieve consensus.

6.5. Future Directions

6.5.a. Estimating deer population size

The work of Chapter 4 concluded Random Encounter Model (REM) to be the most consistent and precise population estimate used, at a relatively low effort. Through a sensitivity analysis in Chapter 4, it was identified that the average daily speed of movement and the detection range of the camera are critical variables of REM, with the ability to substantially impact the estimated population density. Thus accurate estimates are essential.

The closeness of the average abundance estimates (correspondence) of the two population sampling methods (Chapter 4) suggested that the average daily speed of movement used (from Pépin et al., 2004) could be representative of red deer in this comparable environment. However, the similarity between REM and Distance Sampling in one year is not enough to be confident or conclusive of this suggestion. Thus, evaluating the average daily speed of movement of this population of red deer should be a focus for further research. If this is performed using Global Positioning System (GPS) collars, then data for average daily speed of movement of this red deer population can be calculated to calibrate REM and further research can be conducted on how the deer use the landscape. An evaluation on how far deer migrate across landscapes would also be of high value for deer management through evidence for strategic culling locations to improve efficiency in management action.

As a model for estimating wildlife population density, REM is regularly used across a wide range of species (e.g. Garrote, et al., 2021; Kavčić et al., 2021; Logan, 2018; Miura et al., 2022) and is considered a reliable tool for monitoring wildlife with trail cameras (Palencia et al., 2022). However, there are numerous measured factors that are required for REM, including average daily speed of movement and average group size of the deer population. The Random Encounter and Staying Time (REST) is a comparable model considered to be highly effective in woodlands and with low gregarious species (Nakashima et al., 2018). Both REM and REST require independent assessments of average group size. A key advantage of REST over REM is the lack of requirement for average daily movement speed, avoiding the need for GPS collaring of wild deer (e.g. Smith, 2025) or the introduction of landscape markers that may influence wildlife behaviour when estimating average speed of movement from trail cameras. The performance of REST declines for gregarious species because grouping behaviour biases encounter rates and staying times toward social aggregation rather than individual movement, an effect exacerbated in open range landscapes by increased visibility and homogeneous space use. As such, a fully calibrated REM (including average daily speed of movement) and REST should be directly compared and evaluated for correspondence of estimates, precision, and effort to perform.

6.5.b. Deer impact assessments

Chapter 5 demonstrated that current commonly used deer impact assessments (DAI and Putman Method) have met the threshold of being validated indexed methods (see Moriarty et al., 2018), which has increased confidence in their ability to reliably index impacts. However, the areas surveyed in Chapter 5 did not present any high impact areas (DAI and Putman Method) or low-moderate areas (DAI). As such, the methods cannot be considered fully validated, thus further evaluations should be performed to incorporate areas of higher and lower impacts than those already studied (both methods) to ensure the methods are consistent across their full range of values. Further, the areas surveyed here only enabled heather and coarse grassland indicators to be assessed by the Putman Method, thus the untested indicator habitats (smooth grassland and blanket bog) should be evaluated using a similar approach to Chapter 5.

6.6. Recommendations

The work of this thesis explored a broad range of ecological monitoring tools and an engagement of stakeholders across a landscape to build consensus and co-create a framework for adaptive deer management decision making. To maintain important cultural landscapes, the interconnection of ecological health, cultural heritage and human livelihoods must be considered in decision-making. The values of people towards the landscape, therefore, was an important element of the decision-making processes. Key decisions within the co-created framework were to continue to assess the impacts of deer to key habitats across the landscape using appropriate available methodologies, to monitor the population size and trends using reliable data, and to evaluate the impact of deer management decision-making on meeting the co-created management objectives (Figure 3.2). These decisions were all framed around finding the balance between complexity of the methods/analyses of data and using reliable and accurate data by which to make decisions. To find an objective recommendation between methods being "user-friendly" for practitioners (Maffey et al., 2016; Månsson et al., 2023; Palmer et al., 2021), and "high quality" (Figure 3.2) information to inform decision-making, the following recommendations can be made:

6.6.a. Habitat Impact Assessment

The impacts of herbivores can be surveyed using the DAI and Putman Methods for woodlands and upland habitats respectively. These methods both meet the requirements of being user-friendly, and as a result of the work in Chapter 5 they have met the basic requirements of validation for index methods. At a broad scale (low-moderate-high), the methods are consistent at low to moderate-high indexes, but more work is required to ensure this is consistent across their full range of values before they can be considered fully validated methods. Further the use of few categories limits the ability of index methods to detect small temporal changes. Therefore, monitoring will only be effective at detecting changes that cross index categories, reducing the capability of deer managers to identify temporal trends of fine-scale change to impact that do not change index categories. Without identifying trends on a fine scale, it will not be possible to pre-emptively adjust management actions to prevent a categorical index change.

6.6.b. Estimating population size

The most precise estimates of population size can be only ascribed to sampling methods; in this case, the use of trail cameras analysed by REM. The distribution and use of trail cameras is not overly complex but requires deliberate effort to place cameras systematically and with a representative distribution across the landscape. However, based on the recommendations of Maffey et al. (2016) and Palmer et al. (2021), such effort and perceived complexity of analyses may be off-putting to practitioners. To counter this perception, the analyses can be simplified to inputting key data into pre-prepared excel spreadsheets that automatically run the models and provide immediate results.

In contrast, the use of drones may be appealing for relative simplicity in their use and ability to survey large sites in a relatively short period of time. But the inconsistency of minimum population estimates across two years from drone surveys demonstrated the unreliability of surveying landscape-scale animal populations over short temporal scales. Further the drone census counts do not account for changes in detection probability across survey areas, are not systematically surveyed, and are open to biased sampling. Therefore, drones for census, as used in this project, cannot be recommended.

6.7. Conclusion

The work of this thesis has demonstrated an example of best practice in using an interdisciplinary approach, combining ecological surveys with stakeholder engagement for a holistic overview, to achieve consensus for collaborative, landscape-scale adaptive deer impact management.

Using a Delphi process, a facilitated bottom-up, holistic, approach to building consensus in adaptive impact management resulted in co-created decision-making framework as a mitigation to a human-wildlife conflict (Chapter 3). This was achieved by facilitating stakeholders to equitably discuss, define, and create a vision, landscape objectives, and a decision-making framework to mitigate their defined conflict. To improve adaptive management decision-making, evaluations can made from indexed impact assessment methods and use precise estimates of population abundance (such as using REM).

Human-wildlife conflicts are a key factor in increased risk of extinction to wildlife populations, and the associated human-human conflicts are a significant hinderance to developing effective and lasting mitigations and/or solutions. Using an interdisciplinary approach, bringing together the disciplines of ecology and stakeholder elicitation, to create this adaptive impact management framework further facilitated the evaluation of decisions made by participants to enable evidence-based decision-making. By then equitably facilitating discussion across a stakeholder community to holistically review conflicts, such as with a Delphi process, this PhD has demonstrated that consensus-based co-created mitigation strategies for human-wildlife conflict is achievable.

References

Abrahms, B., 2021. Human-wildlife conflict under climate change. *Science*, **373**(6554), pp.484-485.

Adloo, S., Green, G.T. and Boley, B.B., 2023. Measuring "iconicism" through the iconic species scale. *Society & Natural Resources*, **36**(1), pp.1-19.

Ahmad, F., Mori, T., Rehan, M., Bosso, L. and Kabir, M., 2024. Applying a random encounter model to estimate the asiatic black bear (*Ursus thibetanus*) density from camera traps in the hindu raj mountains, Pakistan. *Biology*, **13**(5), p.341.

Ahmed, S.S., Cohen, J. and Anastasopoulos, P.C., 2021. A correlated random parameters with heterogeneity in means approach of deer-vehicle collisions and resulting injury-severities. *Analytic methods in accident research*, **30**, p.100160.

Albon, S.D., Brewer, M.J., O'Brien, S., Nolan, A.J. and Cope, D., 2007. Quantifying the grazing impacts associated with different herbivores on rangelands. *Journal of Applied Ecology*, **44**(6), pp.1176-1187.

Allen, A.M. and Singh, N.J., 2016. Linking movement ecology with wildlife management and conservation. *Frontiers in Ecology and Evolution*, **3**, p.155.

Allen, B., Abraham, A., Arlinghaus, R., Belant, J., Blumstein, D.T., Bobier, C., Bodenchuk, M., Clauss, M., Dawson, S., Derbyshire, S. and Ferreira, S., 2025. Ethical arguments that support intentional animal killing. *Frontiers in Ecology and Evolution*, **13**, p.1684894.

Amit, R. and Jacobson, S.K. 2018. Participatory development of incentives to coexist with jaguars and pumas. *Conservation Biology*. **32**(4), pp.938-948.

Amos, M., Baxter, G., Finch, N., Lisle, A. and Murray, P., 2014. I just want to count them! Considerations when choosing a deer population monitoring method. *Wildlife Biology*, **20**(6), pp.362-370.

Andersen, R., Lund, E., Solberg, E.J. and Saether, B.E. 2010. Ungulates and their management in Norway. In: Apollonio, M., Andersen, R. and Putman, R. eds. *European ungulates and their management in the 21st century*. Cambridge University Press. pp.14-36.

Apollonio, M., Andersen, R. and Putman, R. eds. 2010. *European ungulates and their management in the 21st century*. Cambridge University Press.

Ara, M., Barbeito, I., Kalén, C. and Nilsson, U., 2022. Regeneration failure of Scots pine changes the species composition of young forests. *Scandinavian Journal of Forest Research*, **37**(1), pp.14-22.

Armstrong, H., Black, B., Holl, K. and Thompson, R., 2023. *The woodland herbivore impact assessment method user guide.* Available from: https://www.researchgate.net/publication/370693805_Woodland_Herbivore_Impact_Assessment_Guide_version_5th_April_2023

Baiser, B., Lockwood, J.L., La Puma, D. and Aronson, M.F., 2008. A perfect storm: two ecosystem engineers interact to degrade deciduous forests of New Jersey. *Biological Invasions*, **10**(6), pp.785-795.

Baker, C.J., Campbell, M.A., Udyawer, V., Kopf, R.K. and Campbell, H.A. 2024. The influence of crocodile density on the prevalence of human attacks. *People and Nature*, **6**(5), pp.1922-1932.

Balestrieri, A., Ruiz-González, A., Vergara, M., Capelli, E., Tirozzi, P., Alfino, S., Minuti, G., Prigioni, C. and Saino, N., 2016. Pine marten density in lowland riparian woods: a test of the Random Encounter Model based on genetic data. *Mammalian Biology*, **81**(5), pp.439-446.

Barlow, A.C., Ahmad, I. and Smith, J.L. 2013. Profiling tigers (*Panthera tigris*) to formulate management responses to human-killing in the Bangladesh Sundarbans. *Wildlife Biology in Practice*. **9**(2), pp.30-39.

Barnes, R.F., 2001. How reliable are dung counts for estimating elephant numbers?. *African Journal of Ecology*, **39**(1), pp.1-9.

Baruzzi, C. and Krofel, M., 2017. Friends or foes? Importance of wild ungulates as ecosystem engineers for amphibian communities. *North-Western Journal of Zoology*, **13**(2).

Bashan, D., Colléony, A. and Shwartz, A., 2021. Urban versus rural? The effects of residential status on species identification skills and connection to nature. *People and Nature*, **3**(2), pp.347-358.

Bates, D., Mächler, M., Bolker, B. and Walker, S., 2015. Fitting linear mixed-effects models using lme4. *Journal of statistical software*, **67**(1), pp.1-48.

Bavin, D., MacPherson, J., Denman, H., Crowley, S.L. and McDonald, R.A. 2020. Using Q-methodology to understand stakeholder perspectives on a carnivore translocation. *People and Nature*. **2**(4), pp.1117-1130

Beaver, J.T., Baldwin, R.W., Messinger, M., Newbolt, C.H., Ditchkoff, S.S. and Silman, M.R., 2020. Evaluating the use of drones equipped with thermal sensors as an effective method for estimating wildlife. *Wildlife Society Bulletin*, **44**(2), pp.434-443.

Bender, L.C., Myers, W.L. and Gould, W.R., 2003. Comparison of helicopter and ground surveys for North American elk Cervus elaphus and mule deer Odocoileus hemionus population composition. *Wildlife Biology*, **9**(3), pp.199-205.

Benedetti-Cecchi, L., 2003. The importance of the variance around the mean effect size of ecological processes. *Ecology*, **84**(9), pp.2335-2346.

Bennett, N.J., Whitty, T.S., Finkbeiner, E., Pittman, J., Bassett, H., Gelcich, S. and Allison, E.H., 2018. Environmental stewardship: A conceptual review and analytical framework. *Environmental Management*, **61**(4), pp.597-614.

Bergvall, U.A., Morellet, N., Kjellander, P., Rauset, G.R., Groeve, J.D., Borowik, T., Brieger, F., Gehr, B., Heurich, M., Hewison, A.M. and Kröschel, M., 2021. Settle down! Ranging behaviour responses of roe deer to different capture and release methods. *Animals*, **11**(11), p.3299.

Beringer, J., Hansen, L.P., Wilding, W., Fischer, J. and Sheriff, S.L., 1996. Factors affecting capture myopathy in white-tailed deer. *The Journal of wildlife management*, **60**(2), pp.373-380.

Bertollo, P., 2001. Assessing landscape health: a case study from northeastern Italy. *Environmental Management*, **27**(3), pp.349-365.

Bhatia, S., Redpath, S.M., Suryawanshi, K. and Mishra, C. 2020. Beyond conflict: exploring the spectrum of human–wildlife interactions and their underlying mechanisms. *Oryx*. **54**(5), pp.621-628.

Bieling, C., Eser, U. and Plieninger, T., 2020. Towards a better understanding of values in sustainability transformations: ethical perspectives on landscape stewardship. *Ecosystems and People*, **16**(1), pp.188-196.

Bíl, M., Andrášik, R., Bartonička, T., Křivánková, Z. and Sedoník, J., 2018. An evaluation of odor repellent effectiveness in prevention of wildlife-vehicle collisions. *Journal of Environmental Management*, **205**, pp.209-214.

Bissonette, J.A., Kassar, C.A. and Cook, L.J., 2008. Assessment of costs associated with deer–vehicle collisions: human death and injury, vehicle damage, and deer loss. *Human-Wildlife Conflicts*, **2**(1), pp.17-27.

Bissonette, J.A. and Rosa, S., 2012. An evaluation of a mitigation strategy for deer-vehicle collisions. Wildlife Biology, **18**(4), pp.414-423.

Blossey, B., Curtis, P., Boulanger, J. and Dávalos, A., 2019. Red oak seedlings as indicators of deer browse pressure: Gauging the outcome of different white-tailed deer management approaches. *Ecology and Evolution*, **9**(23), pp.13085-13103.

Bobek, B., Perzanowski, K. and Zieliński, J., 1986. Red deer population census in mountains: testing of an alternative method. *Acta Theriologica*, **31**(31), pp.423-431.

Bobrowski, M., Gillich, B. and Stolter, C., 2020. Nothing else matters? Food as a driving factor of habitat use by red and roe deer in winter?. *Wildlife Biology*, **2020**(4), pp.1-9.

Bödeker, K., Ammer, C., Knoke, T. and Heurich, M. 2021. Determining statistically robust changes in ungulate browsing pressure as a basis for adaptive wildlife management. *Forests*. **12**(8), p.1030.

Bolam, F.C., Grainger, M.J., Mengersen, K.L., Stewart, G.B., Sutherland, W.J., Runge, M.C. and McGowan, P.J., 2019. Using the value of information to improve conservation decision making. *Biological Reviews*, **94**(2), pp.629-647.

Bollen, M., Palencia, P., Vicente, J., Acevedo, P., Del Río, L., Neyens, T., Beenaerts, N. and Casaer, J., 2023. Assessing trends in population size of three unmarked species: A comparison of a multi-species N-mixture model and random encounter models. *Ecology and Evolution*, **13**(10), p.e10595.

Borkowski, J., Dobrowolska, D., Dąbrowski, W., Banul, R. and Załuski, D., 2017. Young conifer stands form a deer browsing refuge for an oak admixture: silvicultural implications for forest regeneration under herbivore pressure. *European Journal of Forest Research*, **136**, pp.787-800.

Boshoff, M., Dissegna, A., Cerizza, A.F., Laner, L., Indovina, A.E., Biasetti, P., Da Re, R., Mascarello, G. and De Mori, B., Wildlife Management and Conservation in South Africa: Informing Legislative Reform through Expert Consultation using the Policy Delphi Methodology. *Frontiers in Veterinary Science*, **12**, p.1549222.

Braun, V. and Clarke, V. 2006., Using thematic analysis in psychology. *Qualitative Research in Psychology*. **3**(2), pp.77-101.

Breed, D., Meyer, L.C., Steyl, J.C., Goddard, A., Burroughs, R. and Kohn, T.A., 2019. Conserving wildlife in a changing world: Understanding capture myopathy—A malignant outcome of stress during capture and translocation. *Conservation Physiology*, **7**(1), p.27.

Briones, M.J.I., 2024. Special feature on ecosystem engineers: Cross-scale and cross-system perspectives. *Functional Ecology*, **38**(1), pp.4-7.

Brown, H.O., Jacobson, S.K., Cockrell, M., Sutt, J., Allen, K. and Copeland, A., 2021. A five-step stakeholder communication plan for more effective natural resource management. *The Journal of Extension*, **59**(4), p.6.

Brunetta, G. and Voghera, A., 2008. Evaluating landscape for shared values: Tools, principles, and methods. *Landscape Research*, **33**(1), pp.71-87.

Brunton, E.A., Srivastava, S.K. and Burnett, S., 2018. Spatial ecology of an urban eastern grey kangaroo (*Macropus giganteus*) population: local decline driven by kangaroo–vehicle collisions. *Wildlife Research*, **45**(8), pp.685-695.

Buckland, S.T., Rexstad, E.A., Marques, T.A. and Oedekoven, C.S., 2015. *Distance sampling: methods and applications*. New York: Springer.

Buckland, S.T., Borchers, D.L., Marques, T.A. and Fewster, R.M., 2023. Wildlife population assessment: changing priorities driven by technological advances. *Journal of Statistical Theory and Practice*, **17**(2), p.20.

Butler, A., 2016. Dynamics of integrating landscape values in landscape character assessment: the hidden dominance of the objective outsider. *Landscape Research*, **41**(2), pp. 239-252.

Burton, A.C., Neilson, E., Moreira, D., Ladle, A., Steenweg, R., Fisher, J.T., Bayne, E. and Boutin, S., 2015. Wildlife camera trapping: a review and recommendations for linking surveys to ecological processes. *Journal of applied ecology*, **52**(3), pp.675-685.

Byg, A., Martin-Ortega, J., Glenk, K. and Novo, P., 2017. Conservation in the face of ambivalent public perceptions–The case of peatlands as 'the good, the bad and the ugly'. *Biological Conservation*, **206**, pp.181-189.

Byg, A., Novo, P. and Kyle, C., 2023. Caring for Cinderella—Perceptions and experiences of peatland restoration in Scotland. *People and Nature*, **5**(2), pp.302-312.

Byrd, E., Lee, J.G. and Widmar, N.J.O., 2017. Perceptions of hunting and hunters by US respondents. *Animals*, **7**(11), p.83.

Campbell-Palmer, R., Rosell, F., Naylor, A., Cole, G., Mota, S., Brown, D., Fraser, M., Pizzi, R., Elliott, M., Wilson, K. and Gaywood, M. 2021. Eurasian beaver (*Castor fiber*) health surveillance in Britain: Assessing a disjunctive reintroduced population. *Veterinary Record*, **188**(8), pp.1-9.

Cap, H., Aulagnier, S. & Deleporte, P. 2002. The phylogeny and behaviour of Cervidae (*Ruminantia Pecora*). *Ethology Ecology & Evolution*. **14**(3), pp.199-216.

Carmichael, J., Cran, A., Hrvatin, F. and Matthews, J., 2023. "We are stewards and caretakers of the land, not exploiters of resources": A qualitative study exploring Canadian farmers' perceptions of environmental sustainability in agriculture. *PLoS One*, **18**(8), p.e0290114.

Carpio, A.J., Apollonio, M. and Acevedo, P. 2021. Wild ungulate overabundance in Europe: contexts, causes, monitoring and management recommendations. *Mammal Review*, **51**(1), pp.95-108.

Carter, N.H. and Linnell, J.D. 2016. Co-adaptation is key to coexisting with large carnivores. *Trends in Ecology & Evolution*, **31**(8), pp.575-578.

Castillo, C.P., Barranco, R.R., Curtale, R., Kompil, M., Jacobs-Crisioni, C., Rodriguez, S.V., Aurambout, J.P., e Silva, F.B., Sulis, P. and Auteri, D., 2024. Are remote rural areas in Europe remarkable? Challenges and opportunities. *Journal of Rural Studies*, **105**, p.103180.

Červený, J., Krojerová-Prokešová, J., Kušta, T. and Koubek, P. 2019. The change in the attitudes of Czech hunters towards Eurasian lynx: is poaching restricting lynx population growth? *Journal for Nature Conservation*. **47**, pp.28-37.

Champagne, E., Raymond, P., Royo, A.A., Speed, J.D., Tremblay, J.P. and Côté, S.D., 2021. A review of ungulate impacts on the success of climate-adapted forest management strategies. *Current Forestry Reports*, **7**(4), pp.305-320.

Chan, K.M., Balvanera, P., Benessaiah, K., Chapman, M., Díaz, S., Gómez-Baggethun, E., Gould, R., Hannahs, N., Jax, K., Klain, S. and Luck, G.W., 2016. Why protect nature? Rethinking values and the environment. *Proceedings of the National Academy of Sciences*, **113**(6), pp.1462-1465.

Charco, J., Perea, R., Gil, L. and Nanos, N., 2016. Impact of deer rubbing on pine forests: implications for conservation and management of Pinus pinaster populations. *European Journal of Forest Research*, **135**(4), pp.719-729.

Chrétien, L.P., Théau, J. and Menard, P., 2015. Wildlife multispecies remote sensing using visible and thermal infrared imagery acquired from an unmanned aerial vehicle (UAV). *The International Archives of the Photogrammetry, Remote Sensing and Spatial Information Sciences*, **40**, pp.241-248.

Cinque, S., Sjölander‐Lindqvist, A. and Sandström, C. 2022. Frontline bureaucrats in wildlife management: Caught in the dilemma between effectiveness and responsiveness. *Environmental Policy and Governance*. **32**(1), pp.17-28.

Clarke, J.L., Welch, D. and Gordon, I.J., 1995. The influence of vegetation pattern on the grazing of heather moorland by red deer and sheep. I. The location of animals on grass/heather mosaics. *Journal of Applied Ecology*, **32**(1) pp.166-176.

Clutton-Brock, T.H., Guinness, F.E. & Albon, S. 1982. *Red Deer, Behavior and Ecology of Two Sexes*. Chicago: The University of Chicago Press.

Clutton-Brock, T.H., Iason, G.R. & Guinness, F. 1987. Sexual segregation and density-related changes in habitat use in male and female Red deer (*Cervus elaphus*). *Journal of Zoology*. **211**(2), pp.275-289.

Cohen, E., 2014. Recreational hunting: Ethics, experiences and commoditization. *Tourism Recreation Research*, **39**(1), pp.3-17.

Cohen, N.E., Brom, F.W. and Stassen, E.N., 2012. Moral convictions and culling animals: A survey in the Netherlands. *Anthrozoös*, **25**(3), pp.353-367.

Conner, M.M., Wood, M.E., Hubbs, A., Binfet, J., Holland, A.A., Meduna, L.R., Roug, A., Runge, J.P., Nordeen, T.D., Pybus, M.J. and Miller, M.W. 2021. The relationship between harvest management and chronic wasting disease prevalence trends in western mule deer (Odocoileus hemionus) herds. *The Journal of Wildlife Diseases*. **57**(4), pp.831-843.

Corgatelli, G., Mattiello, S., Colombini, S. & Crovetto, G. M., 2019. Impact of red deer (Cervus elaphus) on forage crops in a protected area. *Agricultural Systems*, **169**, pp. 41-48.

Corlatti, L. and Zachos, F.E. 2022. *Terrestrial Cetartiodactyla*. Cham: Springer Nature.

Cooke, A.S., 2007. Deer and damage scores for woodland monitoring. *Deer*, **14**(5), pp. 17-20.

Cooke, A.S., 2009. Classifying the impact of deer in woodland. *Deer*, **14**(10), pp. 35-38.

Coomes, D.A., Allen, R.B., Forsyth, D.M. and Lee, W.G., 2003. Factors preventing the recovery of New Zealand forests following control of invasive deer. *Conservation Biology*, **17**(2), pp.450-459.

Cozzi, M., Prete, C., Viccaro, M. and Romano, S., 2019. Impacts of wildlife on agriculture: a spatial-based analysis and economic assessment for reducing damage. *Natural Resources Research*, **28**(1), pp.15-29.

Cravino, A., Perelló, A. and Brazeiro, A., 2024. Livestock–wildlife interactions: key aspects for reconnecting animal production and wildlife conservation. *Animal Frontiers*, **14**(1), pp.13-19.

Cripps, J.K., Pacioni, C., Scroggie, M.P., Woolnough, A.P. and Ramsey, D.S., 2019. Introduced deer and their potential role in disease transmission to livestock in Australia. *Mammal Review*, **49**(1), pp.60-77.

Croft, S., Ward, A.I., Aegerter, J.N. and Smith, G.C., 2019. Modeling current and potential distributions of mammal species using presence-only data: A case study on British deer. *Ecology and Evolution*, **9**(15), pp.8724-8735.

Crozier, G.K.D. and Schulte-Hostedde, A.I., 2014. The ethical dimensions of wildlife disease management in an evolutionary context. *Evolutionary Applications*, **7**(7), pp.788-798.

Csurgó, B. and Smith, M.K., 2021. The value of cultural ecosystem services in a rural landscape context. *Journal of Rural Studies*, **86**, pp.76-86.

Cukor, J., Vacek, Z., Linda, R., Sharma, R.P. and Vacek, S., 2019. Afforested farmland vs. forestland: Effects of bark stripping by Cervus elaphus and climate on production potential and structure of Picea abies forests. *PloS one*, **14**(8), p.e0221082.

Cumming, G.S., 2011. Spatial resilience: integrating landscape ecology, resilience, and sustainability. *Landscape Ecology*, **26**(7), pp.899-909.

Cusack, J.J., Swanson, A., Coulson, T., Packer, C., Carbone, C., Dickman, A.J., Kosmala, M., Lintott, C. and Rowcliffe, J.M., 2015. Applying a random encounter model to estimate lion density from camera traps in Serengeti National Park, Tanzania. *The Journal of Wildlife Management*, **79**(6), pp.1014-1021.

Dandy, N., Ballantyne, S., Moseley, D., Gill, R. and Quine, C. 2009. *The management of roe deer in peri-urban Scotland*. Forest Research: Farnham, UK.

Dandy, N., Ballantyne, S., Moseley, D., Gill, R., Peace A. and Quine, C. 2011. Preferences for wildlife management methods among the peri-urban public in Scotland. *European Journal of Wildlife Research.* **57**, pp.1213-1221.

Daniels, M.J., 2006. Estimating red deer Cervus elaphus populations: an analysis of variation and cost-effectiveness of counting methods. *Mammal Review*, **36**(3), pp.235-247.

Davic, R.D., 2003. Linking keystone species and functional groups: a new operational definition of the keystone species concept. *Conservation Ecology*, **7**(1), p.11.

Davis, A.J., Farrar, R., Jump, B., Hall, P., Guerrant, T. and Pepin, K.M. 2022. An efficient method of evaluating multiple concurrent management actions on invasive populations. *Ecological Applications*. **32**(6), p.e2623.

Deer Act 1991. (c.54). [Online]. London: HMSO. [Accessed 1st July 2024]. Available from: https://www.legislation.gov.uk/ukpga/1991/54/pdfs/ukpga_19910054_en.pdf

DEFRA. 2018. A Green Future: *Our 25 Year Plan to Improve the Environment*. London: Department for Environment, Food and Rural Affairs.

DeGabriel, J.L., Albon, S.D., Fielding, D.A., Riach, D.J., Westaway, S. and Irvine, R.J., 2011. The presence of sheep leads to increases in plant diversity and reductions in the impact of deer on heather. *Journal of Applied Ecology*, **48**(5), pp.1269-1277.

Delibes-Mateos, M., 2015. Conservation conflicts involving mammals in Europe. *Therya*, **6**(1), pp.123-137.

Delisle, Z.J., Swihart, R.K., Quinby, B.M., Sample, R.D., Kinser-Mcbee, K.J., Tauber, E.K. and Flaherty, E.A., 2022. Density from pellet groups: comparing methods for estimating dung persistence time. *Wildlife Society Bulletin*, **46**(3), p.e1325.

De Marinis, A.M., Chirichella, R. and Apollonio, M., 2022. Common fallow deer *Dama dama* (Linnaeus, 1758). In: Corlatti, L. and Zachos, F.E. eds. *Terrestrial Cetartiodactyla, Handbook of the Mammals of Europe*. Cham: Springer Nature. pp.115-154.

Depraetere, M., Pavoine, S., Jiguet, F., Gasc, A., Duvail, S. and Sueur, J., 2012. Monitoring animal diversity using acoustic indices: Implementation in a temperate woodland. *Ecological Indicators*, **13**(1), pp.46-54.

de Vente, J., Reed, M.S., Stringer, L.C., Valente, S. and Newig, J., 2016. How does the context and design of participatory decision making processes affect their outcomes? Evidence from sustainable land management in global drylands. *Ecology and society*, **21**(2), p.24.

Dickman, A.J., 2010. Complexities of conflict: the importance of considering social factors for effectively resolving human–wildlife conflict. *Animal conservation*, **13**(5), pp.458-466.

Dolman, P., Fuller, R., Gill, R., Hooton, D. and Tabor, R., 2010. Escalating ecological impacts of deer in lowland woodland. *British Wildlife*, **21**(4), p.242.

Donfrancesco, V., Allen, B.L., Appleby, R., Behrendorff, L., Conroy, G., Crowther, M.S., Dickman, C.R., Doherty, T., Fancourt, B.A., Gordon, C.E. and Jackson, S.M. 2023. Understanding conflict among experts working on controversial species: A case study on the Australian dingo. *Conservation Science and Practice*. **5**(3), p.e12900.

Drescher, M. and Warriner, G.K., 2022. Environmental Concerns and Stewardship Behaviors Among Rural Landowners: What Supports Farmers and Non-farmers in Being Good Stewards?. *Frontiers in Sustainable Food Systems*, **6**, p.758426.

Drinkwater, R. 2020. 5 Rules for Efficient Workshop Facilitation. *ITNOW*. **62**(1), pp.62-63.

Duncan, P., Tixier, H., Hofmann, R.R. and Lechner-Doll, M., 1998. Feeding strategies and the physiology of digestion in roe deer. In: Andersen, R., Duncan, P. and Linnell, J.D.C. eds. *The European roe deer: the biology of success.* Oslo: Scandinavian University Press, pp. 91-116.

Duporge, I., Spiegel, M.P., Thomson, E.R., Chapman, T., Lamberth, C., Pond, C., Macdonald, D.W., Wang, T. and Klinck, H., 2021. Determination of optimal flight altitude to minimise acoustic drone disturbance to wildlife using species audiograms. *Methods in Ecology and Evolution*, **12**(11), pp.2196-2207.

Dyal, J.R., Miller, K.V., Cherry, M.J. and D'Angelo, G.J., 2022. White-tailed deer movement in response to helicopter surveys. *Wildlife Society Bulletin*, **46**(5), p.e1383.

Edelblutte, É., Short Gianotti, A.G. and Connors, J.P.C., 2022. Perceptions, concerns, and management of white-tailed deer among municipal officials. *Human Dimensions of Wildlife*, **27**(5), pp.436-456.

Ehrhart, S., Stühlinger, M. and Schraml, U. 2022. The relationship of stakeholders' social identities and wildlife value orientations with attitudes toward red deer management. *Human Dimensions of Wildlife*. **27**(1), pp.69-83.

Eriksson, L., Johansson, M., Månsson, J., Sandström, C., Liljebäck, N. and Elmberg, J., 2024. Novel insights into the cognitive, emotional, and experiential dimensions of stakeholder acceptance of wildlife management. *Scientific Reports*, **14**(1), p.29479.

Eriksson, M., Samuelson, L., Jägrud, L., Mattsson, E., Celander, T., Malmer, A., Bengtsson, K., Johansson, O., Schaaf, N., Svending, O. and Tengberg, A., 2018. Water, Forests, people: the Swedish experience in building resilient landscapes. *Environmental Management*, **62**(1), pp.45-57.

Fadaei, E.S., 2019. *Managing deer: attractants, repellents, invasion modelling and impact assessment*. PhD thesis, Queen's University Belfast.

Fairfax, E. and Westbrook, C., 2024. The Ecology and Evolution of Beavers: Ecosystem Engineers that Ameliorate Climate Change. *Annual Review of Ecology, Evolution, and Systematics*, **55**, pp. 323-345.

Fattorini, N., Lovari, S., Watson, P. and Putman, R. 2020. The scale-dependent effectiveness of wildlife management: A case study on British deer. *Journal of Environmental Management*, **276**, p.111303.

Fedyń, I., Przepióra, F., Sobociński, W., Wyka, J. and Ciach, M., 2022. Eurasian beaver–A semi-aquatic ecosystem engineer rearranges the assemblage of terrestrial mammals in winter. *Science of The Total Environment*, **831**, p.154919.

Ferretti, F., Fattorini, L., Sforzi, A. and Pisani, C., 2016. The use of faeces counts to estimate relative densities of wild boar in a Mediterranean area. *Population Ecology*, **58**(2), pp.329-334.

Field, S.A., O'Connor, P.J., Tyre, A.J. and Possingham, H.P., 2007. Making monitoring meaningful. *Austral Ecology*, **32**(5), pp.485-491.

Filyushkina, A., Strange, N., Löf, M., Ezebilo, E.E. and Boman, M., 2018. Applying the Delphi method to assess impacts of forest management on biodiversity and habitat preservation. *Forest Ecology and Management*, **409**, pp.179-189.

Fischer, A., Kereži, V., Arroyo, B., Mateos-Delibes, M., Tadie, D., Lowassa, A., Krange, O. and Skogen, K., 2013. (De) legitimising hunting–Discourses over the morality of hunting in Europe and eastern Africa. *Land Use Policy*, **32**, pp.261-270.

Fitak, R.R., Antonides, J.D., Baitchman, E.J., Bonaccorso, E., Braun, J., Kubiski, S., Chiu, E., Fagre, A.C., Gagne, R.B., Lee, J.S. and Malmberg, J.L., 2019. The expectations and challenges of wildlife disease research in the era of genomics:

Forecasting with a horizon scan-like exercise. *Journal of Heredity*, **110**(3), pp.261-274.

Forestry and Land Scotland. 2021. *Deer numbers placing unprecedented pressure on environment*. [Online]. [13/03/2025]. Available from: https://forestryandland.gov.scot/news-releases/deer-numbers-placing-unprecedented-pressure-on-environment#:~:text=%E2%80%9CThe%20current%20high%20levels%20of,response%20to%20the%20climate%20emergency.

Forestry Commission. 2022. *Reducing the impact of deer on the natural environment - consultation opens.* [Online]. [13/03/2025]. Available from: https://forestrycommission.blog.gov.uk/2022/08/04/reducing-the-impact-of-deer-on-the-natural-environment-consultation-opens/#:~:text=Evidence%20of%20the%20negative%20effects,per%20year%20to%20deer%20damage

Forestry Commission. 2023. *WS1 Deer Habitat Impact Activity Survey Guide* [Online]. London: Available from: https://assets.publishing.service.gov.uk/media/63f8c4058fa8f527f9834c98/WS1_Deer_Habitat_Impact_Activity_Survey_Guide_.docx

Forestry Commission. 2025. *Management requirements for Woodland Supplement WS1 (deer control)* [Online]. London: Available from: https://www.gov.uk/government/publications/management-requirements-for-woodland-supplement-ws1-deer-control-and-management-operations-note-59/management-requirements-for-woodland-supplement-ws1-deer-control-and-management-operations-note-59

Forsyth, D.M., Barker, R.J., Morriss, G. and Scroggie, M.P., 2007. Modeling the relationship between fecal pellet indices and deer density. *The Journal of Wildlife Management*, **71**(3), pp.964-970.

Forsyth, D.M., Comte, S., Davis, N.E., Bengsen, A.J., Côté, S.D., Hewitt, D.G., Morellet, N. and Mysterud, A., 2022. Methodology matters when estimating deer abundance: a global systematic review and recommendations for improvements. *The Journal of Wildlife Management*, **86**(4), p.e22207.

Fraser, M.D., Theobald, V.J., Griffiths, J.B., Morris, S.M. and Moorby, J.M., 2009. Comparative diet selection by cattle and sheep grazing two contrasting heathland communities. *Agriculture, ecosystems & environment*, **129**(1-3), pp.182-192.

Freeman, S.M., 2015. Development of camera trap methodology in monitoring deer distribution and abundance. PhD thesis, Queen's University Belfast.

Friends of Ilkley Moor. 2026. *Flora and Fauna*. [Online]. [Accessed 4 April 2026]. Available from: https://www.ilkleymoor.org/discovering-ilkley-moor/flora-fauna/

Fryxell, J.M., Sinclair, A.R. & Caughley, G. 2014. *Wildlife ecology, conservation, and management*. Chicester: John Wiley & Sons.

Fukuda, Y., Manolis, C. and Appel, K. 2014. Management of human‐crocodile conflict in the Northern Territory, Australia: Review of crocodile attacks and removal of problem crocodiles. *The Journal of Wildlife Management*, **78**(7), pp.1239-1249.

Fuller, A.K., Linden, D.W. and Royle, J.A., 2016. Management decision making for fisher populations informed by occupancy modeling. *The Journal of Wildlife Management*, **80**(5), pp.794-802.

Fuller, A.K., Decker, D.J., Schiavone, M.V. and Forstchen, A.B., 2020. Ratcheting up rigor in wildlife management decision making. *Wildlife Society Bulletin*, **44**(1), pp.29-41.

Garrote, G., de Ayala, R.P., Álvarez, A., Martín, J.M., Ruiz, M., de Lillo, S. and Simón, M.A., 2021. Improving the random encounter model method to estimate carnivore densities using data generated by conventional camera-trap design. *Oryx*, *55*(1), pp.99-104.

Gelves-Gomez, F., Carter, J., Beilin, R. and Brincat, S. 2024. "Ordering the Wild": How Adaptive Management Is Used to Maintain Nature Like a Postcard. *Society and Natural Resources*. **37**(5), pp.791-808.

Gilbert, L., Maffey, G.L., Ramsay, S.L. and Hester, A.J., 2012. The effect of deer management on the abundance of *Ixodes ricinus* in Scotland. *Ecological Applications*, **22**(2), pp.658-667.

Gilchrist, G., Mallory, M. and Merkel, F., 2005. Can local ecological knowledge contribute to wildlife management? Case studies of migratory birds. *Ecology and Society*, **10**(1), p.20.

Gill, R.M.A., 1992. A review of damage by mammals in north temperate forests: 1. Deer. *Forestry*, **65**, pp.145-169.

Gill, R.M.A. and Beardall, V., 2001. The impact of deer on woodlands: the effects of browsing and seed dispersal on vegetation structure and composition. *Forestry*, **74**(3), pp.209-218.

Gill, R.M.A. and Morgan, G. 2010. The effects of varying deer density on natural regeneration in woodlands in lowland Britain. *Forestry*, **83**(1), pp.53-63.

Glass, J.H. 2011. The power of the research process: co-producing knowledge for sustainable upland estate management in Scotland. Ph.D. thesis, Aberdeen University.

Glass, J.H., Scott, A.J. and Price, M.F. 2013. The power of the process: Co-producing a sustainability assessment toolkit for upland estate management in Scotland. *Land Use Policy*, **30**(1), pp.254-265.

Glass, J.H. 2011. The power of the research process: co-producing knowledge for sustainable upland estate management in Scotland. Ph.D. thesis, Aberdeen University.

Gordon, I.J., 2019. Adopting a utilitarian approach to culling wild animals for conservation in National Parks. *Conservation Science and Practice*, **1**(10), p.e105.

Göttert, T. and Starik, N. 2022. Human–wildlife conflicts across landscapes—General applicability vs. case specificity. *Diversity*. **14**(5), p.380.

Goyes, D.R. 2022. The importance of stories in wildlife management. *Ecological Management and Restoration*. **23**(3), pp.237-243.

Graves, T.A., Yarnall, M.J., Johnston, A.N., Preston, T.M., Chong, G.W., Cole, E.K., Janousek, W.M. and Cross, P.C., 2022. Eyes on the herd: Quantifying ungulate density from satellite, unmanned aerial systems, and GPS collar data. *Ecological Applications*, **32**(5), p.e2600.

Gray, T.N., Grainger, M.J. and Grosu, R., 2020. Conservation decision-making under uncertainty: Identifying when to reintroduce tiger Panthera tigris to Cambodia. *Conservation Science and Practice*, **2**(7), p.e187.

Green, D. and Stowe Jr, J.P., 2000. Quality deer management: ethical and social issues. *Human Dimensions of Wildlife*, **5**(4), pp.62-71.

Green, J.N. and Thorogood, N., 2014. *Qualitative Methods for Health Research*. UK: Sage Publications.

Gresham, A., Healey, J.R., Lawrence, P.J., Eichhorn, M.P. and Shannon, G., 2025. A scoping review of evidence for the effects of seven global deer species on woody vegetation. *Ecological Solutions and Evidence*, **6**(2), p.e70044.

Grossmann, C.M. and Patkó, L., 2024. Did we achieve what we aimed for? Assessing the outcomes of a human–carnivore conflict mitigation and coexistence project in Europe. *Wildlife Biology*, **2024**(6), p.e01270.

Gurnell, J., Clark, M.J., Lurz, P.W., Shirley, M.D. and Rushton, S.P., 2002. Conserving red squirrels (Sciurus vulgaris): mapping and forecasting habitat suitability using a Geographic Information Systems Approach. *Biological Conservation*, **105**(1), pp.53-64.

Haefner, J., 1996. *Modelling Biological Systems. Principles and Applications*. London: Chapman and Hall.

Hanley, N. and Perrings, C., 2019. The economic value of biodiversity. *Annual Review of Resource Economics*, **11**(1), pp.355-375.

Hare, D., Daniels, M. and Blossey, B., 2021. Public perceptions of deer management in Scotland. *Frontiers in Conservation Science*, **2**, p.781546.

Heindorf, C., Schüler, S. and Plieninger, T., 2024. Poetic inquiry to explore the relational values of a transforming peat landscape. *People and Nature*, **6**(3), pp.1189-1205.

Hester, A.J., Gordon, I.J., Baillie, G.J. and Tappin, E., 1999. Foraging behaviour of sheep and red deer within natural heather/grass mosaics. *Journal of Applied Ecology*, **36**(1), pp.133-146.

Hmwe, S.S., Zachos, F.E., Sale, J.B., Rose, H.R. and Hartl, G.B., 2006. Genetic variability and differentiation in red deer (*Cervus elaphus*) from Scotland and England. *Journal of Zoology*, **270**(3), pp.479-487.

Hoare, R.E., 1992. Present and future use of fencing in the management of larger African mammals. *Environmental Conservation*, **19**(2), pp.160-164.

Holl, K. and Armstrong, H., 2014. Deer and livestock impacts on native woodlands in Scotland revisited. *Scottish Forestry*, **68**(2), pp.32-36.

Holloway, C.W., 1967. *The effect of red deer and other animals on regenerated scots pine*. PhD thesis, University of Aberdeen.

Hoover, C., Ostertag, S., Hornby, C., Parker, C., Hansen-Craik, K., Loseto, L. and Pearce, T., 2016. The continued importance of hunting for future Inuit food security. *Solutions*, **7**(4), pp.40-50.

Hopkins, C.R., Bailey, D.M. and Potts, T. 2018. Navigating future uncertainty in marine protected area governance: Lessons from the Scottish MPA network. *Estuarine, Coastal and Shelf Science*. **207**, pp.303-311.

Horsley, S., 2020. *Iconic Species and a Sense of Place*. Ph.D. thesis, University of Georgia.

Hothorn, T., Brandl, R. and Müller, J., 2012. Large-scale model-based assessment of deer-vehicle collision risk. *PloS one*, *7*(2), p.e29510.

Hsu, C.C. and Sandford, B.A. 2007. The Delphi technique: making sense of consensus. *Practical assessment, research, and evaluation*. **12**(1). doi: https://doi.org/10.7275/pdz9-th90

Hu, Z., 2020. What socio-economic and political factors lead to global pesticide dependence? A critical review from a social science perspective. *International Journal of Environmental Research and Public Health*, **17**(21), p.8119.

Huber, N., Vetter, S.G., Evans, A.L., Kjellander, P., Küker, S., Bergvall, U.A. and Arnemo, J.M., 2017. Quantifying capture stress in free ranging European roe deer (Capreolus capreolus). *BMC Veterinary Research*, **13**(1), p.127.

Iijima, H. and Nagaike, T., 2015. Appropriate vegetation indices for measuring the impacts of deer on forest ecosystems. *Ecological Indicators*, **48**, pp.457-463.

IUCN. 2020. What is HWC? [Online]. [Accessed 13 May 2025]. Available from: https://www.hwctf.org/_files/ugd/7acc16_c026ab9ffce44ea7900580771cba1cb4.pdf

IUCN. 2022. Issues Brief: Human-wildlife conflict [Online]. [Accessed 12 June 2025]. Available from: https://iucn.org/resources/issues-brief/human-wildlife-conflict#7043

Jakes, A.F., Jones, P.F., Paige, L.C., Seidler, R.G. and Huijser, M.P., 2018. A fence runs through it: A call for greater attention to the influence of fences on wildlife and ecosystems. *Biological Conservation*, **227**, pp.310-318.

Janječić, M., Corlatti, L., Fattorini, N., Ferretti, F., Biondić, D., Safner, T. and Šprem, N., 2025. Variations in random encounter model density estimates of European mouflon island population. *European Journal of Wildlife Research*, **71**(6), p.131.

Jax, K., Calestani, M., Chan, K.M., Eser, U., Keune, H., Muraca, B., O'Brien, L., Potthast, T., Voget-Kleschin, L. and Wittmer, H., 2018. Caring for nature matters: a relational approach for understanding nature's contributions to human well-being. *Current Opinion in Environmental Sustainability*, **35**, pp.22-29.

Jennions, M.D. and Moeller, A.P., 2002. Publication bias in ecology and evolution: an empirical assessment using the 'trim and fill' method. *Biological reviews*, **77**(2), pp.211-222.

Jhala, Y., Gopal, R., Mathur, V., Ghosh, P., Negi, H.S., Narain, S., Yadav, S.P., Malik, A., Garawad, R. and Qureshi, Q., 2021. Recovery of tigers in India: Critical introspection and potential lessons. *People and Nature*, **3**(2), pp.281-293.

Ji, Y., Wei, X., Liu, F., Li, D. and Li, J., 2022. Spatial-temporal patterns of human-wildlife conflicts under coupled impact of natural and anthropogenic factors in Mt. Gaoligong, western Yunnan, China. *Global Ecology and Conservation*, **40**, p.e02329.

Johnson, B.B. and Horowitz, L.S., 2014. Beliefs about ecological impacts predict deer acceptance capacity and hunting support. *Society & Natural Resources*, **27**(9), pp.915-930.

Jones, C. and Kirk, N. 2018. Shared visions: can community conservation projects' outcomes inform on their likely contributions to national biodiversity goals?. *New Zealand Journal of Ecology*, **42**(2), pp.116-124.

Jørgensen, S., Xu, L. and Costanza, R. eds., 2016. *Handbook of ecological indicators for assessment of ecosystem health*. 2nd Ed. Boca Raton: CRC press.

Jourdain, N.O., Cole, D.J., Ridout, M.S. and Rowcliffe, J.M., 2020. Statistical development of animal density estimation using random encounter modelling. *Journal of Agricultural, Biological and Environmental Statistics*, **25**, pp.148-167.

Kaiser, J., Hahn, T., Krueger, T. and Haase, D. 2025. Investigating conservation performance payments alongside human–wildlife conflicts: The Swedish lynx and wolverine protection policies. *People and Nature,* **00**, pp.1-19

Kaltenborn, B.P. and Linnell, J.D., 2022. The coexistence potential of different wildlife conservation frameworks in a historical perspective. *Frontiers in Conservation Science*, **2**, p.711480.

Kansky, R., Kidd, M. & Knight, A. T. 2016. A wildlife tolerance model and case study for understanding human wildlife conflicts. *Biological Conservation*, **201**, pp. 137-145.

Karki, B.K., Silwal, T. and Paudel, S.R., 2025. Relocating Nepal's Problematic Tigers to Suitable Habitats: A Lens of Management. *Natural Built Social Environment Health*, **1**(1), pp.3-6.

Kavčić, K., Palencia, P., Apollonio, M., Vicente, J. and Šprem, N., 2021. Random encounter model to estimate density of mountain-dwelling ungulate. *European Journal of Wildlife Research*, **67**(5), p.87.

Kéry, M. and Schmidt, B., 2008. Imperfect detection and its consequences for monitoring for conservation. *Community Ecology*, **9**(2), pp.207-216.

Kilpatrick, A.M., Hoyt, J.R., King, R.A., Kaarakka, H.M., Redell, J.A., White, J.P. and Langwig, K.E., 2020. Impact of censusing and research on wildlife populations. *Conservation Science and Practice*, **2**(11), p.e264.

Kilpatrick, H.J. and LaBonte, A.M. 2003. Deer hunting in a residential community: the community's perspective. *Wildlife Society Bulletin*. **31,** 340-348.

Kingsford, R.T., West, R.S., Pedler, R.D., Keith, D.A., Moseby, K.E., Read, J.L., Letnic, M., Leggett, K.E. and Ryall, S.R., 2021. Strategic adaptive management planning—Restoring a desert ecosystem by managing introduced species and native herbivores and reintroducing mammals. *Conservation Science and Practice*, **3**(2), p.e268.

Kirkland, H., Hare, D., Daniels, M., Krofel, M., Rao, S., Chapman, T. and Blossey, B. 2021. Successful deer management in Scotland requires less conflict not more. *Frontiers in Conservation Science*, **2**, p.770303.

Kochprapa, P., Savini, C., Ngoprasert, D., Savini, T. and Gale, G., 2024. Mitigating Human-Elephant Conflict in Southeast Asia. *Tropical Natural History*, **24**, pp.70-83.

König, H.J., Ceaușu, S., Reed, M., Kendall, H., Hemminger, K., Reinke, H., Ostermann‐Miyashita, E.F., Wenz, E., Eufemia, L., Hermanns, T. and Klose, M. 2021. Integrated framework for stakeholder participation: Methods and tools for identifying and addressing human–wildlife conflicts. *Conservation Science and Practice*. **3**(3), p.e399.

König, H.J., Kiffner, C., Kramer‐Schadt, S., Fürst, C., Keuling, O. and Ford, A.T. 2020. Human‐wildlife coexistence in a changing world. *Conservation Biology*. **34**(4), pp.786-794.

Kontogianni, A., Tziritis, I. and Skourtos, M., 2005. Bottom-up environmental decision making taken seriously: integrating stakeholder perceptions into scenarios of environmental change. *Human Ecology Review*, **12**(2), pp.87-95.

Kotulski, Y. and König, A. 2008. Conflicts, crises and challenges: wild boar in the Berlin City – a social, empirical and statistical survey. *Natura Croatica*. **17**(4), 233–246.

Kowalewski, L.K., Chizinski, C.J., Powell, L.A., Pope, K.L. and Pegg, M.A., 2015. Accuracy or precision: implications of sample design and methodology on abundance estimation. *Ecological modelling*, **316**, pp.185-190.

Kueffer, C. and Kull, C.A., 2017. Non-native species and the aesthetics of nature. In Vilà, M. and Hulme, P.E. (eds). *Impact of biological invasions on ecosystem services*. Cham: Springer, pp.311-324

Laing, S.E., Buckland, S.T., Burn, R.W., Lambie, D. and Amphlett, A., 2003. Dung and nest surveys: estimating decay rates. *Journal of Applied Ecology*, **40**(6), pp.1102-1111.

Lamichhane, B.R., Persoon, G.A., Leirs, H., Poudel, S., Subedi, N., Pokheral, C.P., Bhattarai, S., Gotame, P., Mishra, R. and De Iongh, H.H., 2019. Contribution of buffer zone programs to reduce human-wildlife impacts: the case of the Chitwan National Park, Nepal. *Human Ecology*, **47**(1), pp.95-110.

Laundré, J.W., Hernández, L. and Ripple, W.J., 2010. The landscape of fear: ecological implications of being afraid. *Open Ecology Journal*, **3**(3), pp.1-7.

Lederman, Z., Magalhães-Sant'Ana, M. and Voo, T.C., 2021. Stamping out animal culling: From anthropocentrism to one health ethics. *Journal of Agricultural and Environmental Ethics*, **34**(5), p.27.

Lees, K.J., Carmenta, R., Condliffe, I., Gray, A., Marquis, L. and Lenton, T.M., 2023. Protecting peatlands requires understanding stakeholder perceptions and relational values: A case study of peatlands in the Yorkshire Dales. *Ambio*, **52**(7), pp.1282-1296.

Legg, C.J. and Nagy, L., 2006. Why most conservation monitoring is, but need not be, a waste of time. *Journal of Environmental Management*, **78**(2), pp. 194 199.

Levy, P.S. and Lemeshow, S., 2013. *Sampling of populations: methods and applications*. John Wiley & Sons.

Lewis, C.L., Granek, E.F. and Nielsen-Pincus, M., 2019. Assessing local attitudes and perceptions of non-native species to inform management of novel ecosystems. *Biological Invasions*, **21**(3), pp.961-982.

Liberg, O., Bergström, R., Kindlberg, J. and von-Essen, H., 2010. Ungulates and their management in Sweden. In: Apollonio, M., Andersen, R. and Putman, R. eds. *European ungulates and their Management in the 21st Century*. Cambridge: Cambridge University Press. pp. 37-70.

Lindberg, M.S., 2012. A review of designs for capture–mark–recapture studies in discrete time. *Journal of Ornithology*, **152**, pp.355-370.

Linnell, J.D., Cretois, B., Nilsen, E.B., Rolandsen, C.M., Solberg, E.J., Veiberg, V., Kaczensky, P., Van Moorter, B., Panzacchi, M., Rauset, G.R. and Kaltenborn, B. 2020.

The challenges and opportunities of coexisting with wild ungulates in the human-dominated landscapes of Europe's Anthropocene. *Biological Conservation*, **244**, p.108500.

Linstone, H.A., and M. Turoff. 2002. The Delphi Method: Techniques and Applications. *Futures research methodology*. Addison-Wesley.

Lischka, S.A., Riley, S.J. and Rudolph, B.A., 2008. Effects of impact perception on acceptance capacity for white‐tailed deer. *The Journal of Wildlife Management*, **72**(2), pp.502-509.

Littin, K., Fisher, P., Beausoleil, N.J. and Sharp, T., 2014. Welfare aspects of vertebrate pest control and culling: ranking control techniques for humaneness. *Revue scientifique et technique (International Office of Epizootics)*, **33**(1), pp.281-289.

Liukkonen, T., Mykrä, S., Bisi, J. and Kurki, S. 2009. Conflicts and compromises in lynx *Lynx lynx* conservation and management in Finland. *Wildlife Biology*, **15**(2), pp.165-174.

Logan, T.W. 2018. *The Detection and Quantification of Deer Populations for Impact Management on Thorne Moors; Humberhead Peatlands National Nature Reserve.* M.Sc. thesis, University of Hull. Available from: https://hull-repository.worktribe.com/output/4221579

Logan, T.W., Ashton-Butt, A. and Ward, A.I., 2019. Improving daytime detection of deer for surveillance and management. *European Journal of Wildlife Research*, **65**(6), p.83.

Logan, T.W., Ward, A.I. and Hopkins, C.R., 2025. Stakeholder solutions to human-wildlife conflicts: Co-created adaptive impact management for wild red deer, *Cervus elaphus*, in the English Lake District. *Biological Conservation*, **310**, p.111380.

Loker, C.A., Decker, D.J. and Schwager, S.J. 1999. Social acceptability of wildlife management actions in suburban areas: three case studies from New York. *Wildlife Society Bulletin*. **27**, 152-159.

Lorenzini, R., Hewison, M., Gaillard, J-M., Herofalo, L., Rossi, L., Morellet, N., Verheyden, H., Lovari, S., Lister, D.M. and Mattioli, S., 2022. European roe deer *Capreolus capreolus* (Linnaeus, 1758). In: Corlatti, L. and Zachos, F.E. eds. *Terrestrial*

Cetartiodactyla, Handbook of the mammals of Europe. Cham: Springer Nature. pp.165-196.

Lovari, S., Lorenzini, R., Masseti, M., Pereladova, O., Carden, R.F., Brook, S.M. and Mattioli, S. 2018. *Cervus elaphus (errata version published in 2019). The IUCN Red List of Threatened Species 2018: e.T55997072A142404453*. [online]. [Accessed 11 October 2021]. Available from World Wide Web: <https://dx.doi.org/10.2305/IUCN.UK.2018-2.RLTS.T55997072A142404453.en>

Lowe, V.P.W. & Gardiner, A.S. 1975. Hybridization between red deer (Cervus elaphus) and sika deer (Cervus nippon) with particular reference to stocks in NW England. *Journal of Zoology*. **177**(4), pp.553-566.

Lucas, T.C., Moorcroft, E.A., Freeman, R., Rowcliffe, J.M. and Jones, K.E., 2015. A generalised random encounter model for estimating animal density with remote sensor data. *Methods in Ecology and Evolution*, **6**(5), pp.500-509.

Ludwig, S.C., Roos, S., Bubb, D. and Baines, D., 2017. Long-term trends in abundance and breeding success of red grouse and hen harriers in relation to changing management of a Scottish grouse moor. *Wildlife Biology*, **2017**, pp.1-7.

Ludwig, S.C., Aebischer, N.J., Richardson, M., Roos, S., Thompson, D.B., Wilson, J.D. and Baines, D., 2020. Differential responses of heather and red grouse to long-term spatio-temporal variation in sheep grazing. *Biodiversity and Conservation*, **29**(8), pp.2689-2710.

Lynch, A.J., Thompson, L.M., Morton, J.M., Beever, E.A., Clifford, M., Limpinsel, D., Magill, R.T., Magness, D.R., Melvin, T.A., Newman, R.A. and Porath, M.T., 2022. RAD adaptive management for transforming ecosystems. *BioScience*, **72**(1), pp.45-56.

Lyons, J.E., Runge, M.C., Laskowski, H.P. and Kendall, W.L., 2008. Monitoring in the context of structured decision-making and adaptive management. *The Journal of Wildlife Management*, **72**(8), pp.1683-1692.

Ma, D., Abrahms, B., Allgeier, J., Newbold, T., Weeks, B.C. and Carter, N.H., 2024. Global expansion of human-wildlife overlap in the 21st century. *Science Advances*, **10**(34), p.eadp7706.

MacDonald, A.M., Jones, P.F., Hanlon, J.A., Martin, B.H. and Jakes, A.F., 2022. How did the deer cross the fence: An evaluation of wildlife-friendlier fence modifications to facilitate deer movement. *Frontiers in Conservation Science*, **3**, p.991765.

MacDonald, A.M., Stevens, P., Armstrong, H., Immiriz, P. and Reynolds, P. 1998a. *A Guide to Upland Habitats; Surveying Land Management Impacts Volume 1: Background Information and Guidance for Surveyors*. Edinburgh, UK: Scottish Natural Heritage.

MacDonald, A.M., Stevens, P., Armstrong, H., Immiriz, P. and Reynolds, P. 1998b. *A Guide to Upland Habitats Surveying Land Management Impacts. Volume 2: The Field Guide*. Edinburgh, UK: Scottish Natural Heritage.

MacMillan, D.C. and Marshall, K. 2006. The Delphi process – an expert-based approach to ecological modelling in data-poor environments. *Animal Conservation*. **9**(1), pp.11-19.

Madden, F. and McQuinn, B., 2014. Conservation's blind spot: The case for conflict transformation in wildlife conservation. *Biological Conservation*, **178**, pp.97-106.

Maffey, G., Irvine, R.J., Reed, M. and van der Wal, R., 2016. Can digital reinvention of ecological monitoring remove barriers to its adoption by practitioners? A case study of deer management in Scotland. *Journal of environmental management*, **184**, pp.186-195.

Manfredo, M.J., Berl, R.E., Teel, T.L. and Bruskotter, J.T., 2021. Bringing social values to wildlife conservation decisions. *Frontiers in Ecology and the Environment*, **19**(6), pp.355-362.

Manning, A.D., Gordon, I.J. and Ripple, W.J., 2009. Restoring landscapes of fear with wolves in the Scottish Highlands. *Biological Conservation*, **142**(10), pp.2314-2321.

Månsson, J., Eriksson, L., Hodgson, I., Elmberg, J., Bunnefeld, N., Hessel, R., Johansson, M., Liljebäck, N., Nilsson, L., Olsson, C. and Pärt, T., 2023. Understanding and overcoming obstacles in adaptive management. *Trends in ecology & evolution*, **38**(1), pp.55-71.

Marada, P., Cukor, J., Linda, R., Vacek, Z., Vacek, S. and Havránek, F., 2019. Extensive orchards in the agricultural landscape: Effective protection against fraying damage caused by roe deer. *Sustainability*, **11**(13), p.3738.

Martin, J.L., Chamaillé-Jammes, S. and Waller, D.M., 2020. Deer, wolves, and people: costs, benefits and challenges of living together. *Biological Reviews*, **95**(3), pp.782-801.

Martin, J., Vourc'h, G., Bonnot, N., Cargnelutti, B., Chaval, Y., Lourtet, B., Goulard, M., Hoch, T., Plantard, O., Hewison, A.M. and Morellet, N., 2018. Temporal shifts in landscape connectivity for an ecosystem engineer, the roe deer, across a multiple-use landscape. *Landscape ecology*, **33**, pp.937-954.

Martínez-Jauregui, M., Delibes-Mateos, M., Arroyo, B., Glikman, J.A. and Soliño, M., 2023. Beyond rural vs urban differences: A close match in european preferences in some basic wildlife management and conservation principles. *Journal of Environmental Management*, **331**, p.117236.

Massei, G., 2023. Fertility control for wildlife: a European perspective. *Animals*, **13**(3), p.428.

Massei, G., Quy, R.J., Gurney, J. and Cowan, D.P., 2010. Can translocations be used to mitigate human–wildlife conflicts?. *Wildlife Research*, **37**(5), pp.428-439.

Mattioli, S., Zachos, F.E., Rossi, L., Lister, A.M. and Corlatti, L. 2022. Red Deer Cervus elaphus Linnaeus, 1758. In: Corlatti, L. and Zachos, F.E. eds. *Terrestrial Cetartiodactyla, Handbook of the mammals of Europe*. Cham: Springer Nature. pp.51-86.

Mayle, B.A., Peace, A.J., Gill, R.M.A. (1999) *How Many Deer? A Field Guide to Estimating Deer Population Size.* Forestry Commission Field Book 18. HMSO

Mazehan, S.M., Shuib, B.K. and Hashim, H., 2013. Value of rural landscape from public perspectives. *In Proceedings of the International Conference on Social Science Research*, ICSSR (pp. 4-5).

McCarthy, M.A., Moore, A.L., Krauss, J., Morgan, J.W. and Clements, C.F., 2014. Linking indices for biodiversity monitoring to extinction risk theory. *Conservation Biology*, **28**(6), pp.1575-1583.

McCollister, M.F. and Van Manen, F.T., 2010. Effectiveness of wildlife underpasses and fencing to reduce wildlife-vehicle collisions. *The Journal of Wildlife Management*, **74**(8), pp.1722-1731.

McDonald-Madden, E., Baxter, P.W., Fuller, R.A., Martin, T.G., Game, E.T., Montambault, J. and Possingham, H.P., 2010. Monitoring does not always count. *Trends in ecology & evolution*, **25**(10), pp.547-550.

McShea, W.J. and Rappole, J.H., 1992. White-tailed deer as keystone species within forest habitats of Virginia. *Virginia Journal of Science*, **43**(1B), pp.177-186.

Melletti, M. and Focardi, S., 2025. *Deer of the World: Ecology, Conservation and Management*. Cham: Springer Nature.

Melton, C.B., Reside, A.E., Simmonds, J.S., Mcdonald, P.G., Major, R.E., Crates, R., Catterall, C.P., Clarke, M.F., Grey, M.J., Davitt, G. and Ingwersen, D., 2021. Evaluating the evidence of culling a native species for conservation benefits. *Conservation Science and Practice*, **3**(12), p.e549.

Melville, R.C., Tee, L.A. and Rennolls, K., 1983. *Assessment of wildlife damage in forests*. London: Linneys of Mansfield.

Mentges, A., Blowes, S.A., Hodapp, D., Hillebrand, H. and Chase, J.M., 2021. Effects of site-selection bias on estimates of biodiversity change. Conservation Biology, **35**(2), pp.688-698.

Merkle, J.A., Anderson, N.J., Baxley, D.L., Chopp, M., Gigliotti, L.C., Gude, J.A., Harms, T.M., Johnson, H.E., Merrill, E.H., Mitchell, M.S. and Mong, T.W. 2019. A collaborative approach to bridging the gap between wildlife managers and researchers. *The Journal of Wildlife Management*. **83**(8), pp.1644-1651.

Merson, S.D., Dollar, L.J., Johnson, P.J. and Macdonald, D.W., 2019. Retaliatory killing and human perceptions of Madagascar's largest carnivore and livestock predator, the fosa (*Cryptoprocta ferox*). *PloS One*, **14**(3), p.e0213341.

Messmer, T.A., 2000. The emergence of human–wildlife conflict management: turning challenges into opportunities. *International Biodeterioration & Biodegradation*, **45**(3-4), pp.97-102.

Messmer, T.A., 2009. Human–wildlife conflicts: emerging challenges and opportunities. *Human-Wildlife Conflicts*, **3**(1), pp.10-17.

Mikołajczak, K.M., Jones, N., Sandom, C.J., Wynne‐Jones, S., Beardsall, A., Burgelman, S., Ellam, L. and Wheeler, H.C., 2022. Rewilding—The farmers' perspective. Perceptions and attitudinal support for rewilding among the English farming community. *People and Nature*, **4**(6), pp.1435-1449.

Miller, K.K. and Jones, D.N., 2005. Wildlife management in Australasia: perceptions of objectives and priorities. *Wildlife Research*, **32**(4), pp.265-272.

Miller, K.M., Berg, G., Indigenous Knowledge Keepers of Churchill Berg Georgina., Lickers, M., McIvor, N. and Henri, D.A. 2025. Coexistence between people and polar bears supports Indigenous knowledge mobilization in wildlife management and research. *Communications Earth & Environment*, **6**(1), p.74.

Mills, L.S., Soulé, M.E. & Doak, D.F. 1993. The keystone-species concept in ecology and conservation. *BioScience*. **43**(4), pp.219-224.

Mishra, S., 2024. Keystone species and their impact on the ecosystem: Current trends. *Journal of Ecology and Conservation*, **2**(1), pp.1-4.

Miura, K., Mori, T., Ogawa, H., Umano, S., Kato, H., Izumiyama, S. and Niizuma, Y., 2022. Density estimations of the Asiatic black bear: application of the random encounter model. *Journal of Natural History*, **56**(21-24), pp.1123-1138.

Montané, J., Marco, I., Manteca, X., López, J. and Lavín, S., 2002. Delayed acute capture myopathy in three roe deer. *Journal of Veterinary Medicine Series A*, **49**(2), pp.93-98.

Moore, E.K., Iason, G.R., Pemberton, J.M., Bryce, J., Dayton, N., Britton, A.J. and Pakeman, R.J. 2018. Habitat impact assessment detects spatially driven patterns of grazing impacts in habitat mosaics but overestimates damage. *Journal for Nature Conservation*, **45**, pp.20-29.

Morellet, N., Klein, F., Solberg, E. and Andersen, R., 2011. The census and management of populations of ungulates in Europe. In: Putman, R.J., Apollonio, M. and Andersen, R. eds. *Ungulate Management in Europe: Problems and Practices*. Cambridge: Cambridge University Press, pp.106-143.

Morgan, D.G., Gibbens, J.R., Conway, E.T., Hepworth, G. and Clough, J., 2024. Wildlife density estimation by distance sampling: A novel technique with movement compensation. *PloS one*, **19**(10), p.e0310020.

Moriarty, P.E., Hodgson, E.E., Froehlich, H.E., Hennessey, S.M., Marshall, K.N., Oken, K.L., Siple, M.C., Ko, S., Koehn, L.E., Pierce, B.D. and Stawitz, C.C., 2018. The need for validation of ecological indices. *Ecological Indicators*, **84**, pp.546-552.

Mukherjee, N., Huge, J., Sutherland, W.J., McNeill, J., Van Opstal, M., Dahdouh-Guebas, F. and Koedam, N. 2015. The Delphi technique in ecology and biological conservation: applications and guidelines. *Methods in Ecology and Evolution*. **6**(9), pp.1097-1109.

Mukherjee, N., Zabala, A., Huge, J., Nyumba, T.O., Adem Esmail, B. and Sutherland, W.J., 2018. Comparison of techniques for eliciting views and judgements in decision‐making. *Methods in Ecology and Evolution*, **9**(1), pp.54-63.

Müller, A., Dahm, M., Bøcher, P.K., Root-Bernstein, M. and Svenning, J.C., 2017. Large herbivores in novel ecosystems-Habitat selection by red deer (Cervus elaphus) in a former brown-coal mining area. *PLoS One*, **12**(5), p. e0177431.

Murray, M.H., Becker, D.J., Hall, R.J. and Hernandez, S.M., 2016. Wildlife health and supplemental feeding: a review and management recommendations. *Biological Conservation*, **204**, pp.163-174.

Nagy-Reis, M., Reimer, J.R., Lewis, M.A., Jensen, W.F. and Boyce, M.S. 2021. Aligning population models with data: Adaptive management for big game harvests. *Global Ecology and Conservation*. **26**, p.e01501.

Naha, D., Dash, S.K., Chettri, A., Roy, A. and Sathyakumar, S. 2020. Elephants in the neighborhood: Patterns of crop-raiding by Asian elephants within a fragmented landscape of Eastern India. *PeerJ*. **8**, p.e9399.

Nakashima, Y., Fukasawa, K. and Samejima, H., 2018. Estimating animal density without individual recognition using information derivable exclusively from camera traps. *Journal of Applied Ecology*, **55**(2), pp.735-744.

National Research Council, 1992. *Restoration of aquatic ecosystems: science, technology, and public policy*. Washington DC: National Academies Press.

NatureScot, 2024. *Deer Best Practice Guidance for Scotland*. [Online]. [Accessed 21 November 2024]. Available from https://bestpracticeguides.org.uk/impacts/

NatureScot. 2024. *Deer management: Policy, Guidance and Best Practice*. [Online]. [Accessed 19th March 2025]. Available from: https://www.nature.scot/professional-advice/land-and-sea-management/managing-wildlife/managing-scotlands-wild-deer/deer-management-policy-guidance-and-best-practice

Naylor, R., Manley, W., Maye, D., Enticott, G., Ilbery, B. and Hamilton‐Webb, A., 2017. The framing of public knowledge controversies in the media: a comparative analysis of the portrayal of badger vaccination in the English national, regional and farming press. *Sociologia Ruralis*, **57**(1), pp.3-22.

Nelli, L., Langbein, J., Watson, P. and Putman, R., 2018. Mapping risk: Quantifying and predicting the risk of deer-vehicle collisions on major roads in England. *Mammalian Biology*, **91**(1), pp.71-78.

Nelson, B.W., Walters, C.J., Trites, A.W. and McAllister, M.K. 2023. Comparing lethal and non‐lethal methods of active population control for harbor seals in British Columbia. *The Journal of Wildlife Management*. **84**(5), p.e22400.

Ngene, S., Ihwagi, F., Nzisa, M., Mukeka, J., Njumbi, S. and Omondi, P., 2011. Total aerial census of elephants and other large mammals in the Tsavo-Mkomazi ecosystem. *Kenya Wildlife Service report*.

Nguyen, L.B., Chen, H.C., Seekings, T.B.W., Dhungana, N., Chen, C.C. and Lee, C.H., 2024. Integrated adaptation strategies for human–leopard cat coexistence management in Taiwan. *Sustainability*, **16**(10), p.4031.

Nichols, J.D. and Williams, B.K., 2006. Monitoring for conservation. *Trends in ecology & evolution*, **21**(12), pp.668-673.

Niemiec, R.M., Quartuch, M., Gonzalez, M., Champine, V. and Mertens, A., 2025. Social outcomes associated with a public and stakeholder engagement process for wolf reintroduction. *Conservation Science and Practice*, p.e70058.

Nilsen, E.B., Milner-Gulland, E.J., Schofield, L., Mysterud, A., Stenseth, N.C. and Coulson, T., 2007. Wolf reintroduction to Scotland: public attitudes and consequences for red deer management. *Proceedings of the Royal Society B: Biological Sciences*, **274**(1612), pp.995-1003.

Nyhus, P.J. 2016. Human–wildlife conflict and coexistence. *Annual review of environment and resources*, **41**(1), pp.143-171.

Ogden, L.E., 2023. Supplemental feeding: examining its place in the wildlife conservation toolbox. *Bioscience*, **73**(2), pp.77-84.

Olsen, C. S. 2022. Co-Creation Beyond Humans: The Arts of Multispecies Placemaking. *Urban Planning,* **7**(2), pp. 315-325.

O'Neill, H.M., Twiss, S.D., Stephens, P.A., Mason, T.H., Ryrholm, N. and Burman, J., 2022. The importance of direct and indirect trophic interactions in determining the presence of a locally rare day-flying moth. *Oecologia*, **198**(2), pp. 531-542.

Oom, S.P., Sibbald, A.M., Hester, A.J., Miller, D.R. and Legg, C.J., 2008. Impacts of sheep grazing a complex vegetation mosaic: relating behaviour to vegetation change. *Agriculture, ecosystems & environment*, **124**(3-4), pp.219-228.

Owen, M.A., Swaisgood, R.R. and Blumstein, D.T., 2017. Contextual influences on animal decision-making: significance for behavior-based wildlife conservation and management. *Integrative Zoology*, **12**(1), pp.32-48.

Pagon, N., Grignolio, S., Pipia, A., Bongi, P., Bertolucci, C. and Apollonio, M., 2013. Seasonal variation of activity patterns in roe deer in a temperate forested area. *Chronobiology international*, **30**(6), pp.772-785.

Paine, R.T., 1969. A note on trophic complexity and community stability. *The American Naturalist*, **103**(929), pp.91-93.

Paine, R.T., 1995. A conversation on refining the concept of keystone species. *Conservation Biology*, **9**, pp.962-964.

Palencia, P., Barroso, P., Vicente, J., Hofmeester, T.R., Ferreres, J. and Acevedo, P., 2022. Random encounter model is a reliable method for estimating population density of multiple species using camera traps. *Remote Sensing in Ecology and Conservation*, **8**(5), pp.670-682.

Palmer, G., 2014. *Deer in Britain: population spread and the implications for biodiversity*. Ph.D. thesis, Durham University. Available from: https://etheses.dur.ac.uk/10597/

Palmer, A., Reynolds, S.J., Lane, J., Dickey, R. and Greenhough, B., 2021. Getting to grips with wildlife research by citizen scientists: What role for regulation?. *People and Nature*, **3**(1), pp.4-16.

Pan, Y., Piras, F., Wang, Z., Lai, M., Agnoletti, M. and Santoro, A., 2025. A methodological framework for identifying traditional rural landscapes based on environmental, cultural, and socio-economic indicators-the case study of China. *Scientific Reports*, **15**(1), p.18650.

Paras, C., Michaud, T. and Hoffman, M., 2022. Sustaining New England's iconic tourism landscapes: An exploratory study to examine perceptions of value from farmers and fishermen. *Journal of Agriculture, Food Systems, and Community Development*, **12**(1), pp.141-156.

Pépin, D., Adrados, C., Mann, C. and Janeau, G., 2004. Assessing real daily distance travelled by ungulates using differential GPS locations. *Journal of Mammalogy*, **85**(4), pp.774-780.

Pépin, D., Renaud, P.C., Boscardin, Y., Goulard, M., Mallet, C., Anglard, F. and Ballon, P., 2006. Relative impact of browsing by red deer on mixed coniferous and broad-leaved seedlings—an enclosure-based experiment. *Forest Ecology and Management*, **222**(1-3), pp.302-313.

Pépin, D., Morellet, N. and Goulard, M., 2009. Seasonal and daily walking activity patterns of free-ranging adult red deer (*Cervus elaphus*) at the individual level. *European Journal of Wildlife Research*, **55**(5), pp. 479-486

Pepper, S., Barbour, A., and Glass, J. 2019. *The Management of Wild Deer in Scotland.* [Online]. [Accessed 17th March 2025]. Available from: https://www.gov.scot/publications/management-wild-deer-scotland/pages/5/

Pérez-Espona, S., Pemberton, J.M. and Putman, R., 2009. Red and sika deer in the British Isles, current management issues and management policy. *Mammalian Biology*, **74**(4), pp.247-262.

Pérez-Espona, S., Hall, R.J., Pérez-Barbería, F.J., Glass, B.C., Ward, J.F. and Pemberton, J.M., 2013. The impact of past introductions on an iconic and economically important species, the red deer of Scotland. *Journal of Heredity*, **104**(1), pp.14-22.

Petrovan, S.O., Ward, A.I. and Wheeler, P., 2011. Detectability counts when assessing populations for biodiversity targets. *PloS one*, **6**(9), p.e24206.

Pettigrew, P., Sigouin, D. and St-Laurent, M.H., 2021. Testing the precision and sensitivity of density estimates obtained with a camera-trap method revealed limitations and opportunities. *Ecology and Evolution*, **11**(12), pp.7879-7889.

Pfeffer, S.E., Spitzer, R., Allen, A.M., Hofmeester, T.R., Ericsson, G., Widemo, F., Singh, N.J. and Cromsigt, J.P., 2018. Pictures or pellets? Comparing camera trapping and dung counts as methods for estimating population densities of ungulates. *Remote Sensing in Ecology and Conservation*, **4**(2), pp.173-183.

Pierson, T.G. and deCalesta, D.S., 2015. Methodology for estimating deer browsing impact. *Human-Wildlife Interactions*, **9**(1), pp. 67-77.

Pimid, M., Mohd, N.M.R., Krishnan, K.T., Chambers, G.K., Ahmad, A.G. and Perijin, J. 2022. Understanding Social Dimensions in Wildlife Conservation: Multiple Stakeholder Views. *Animals*. **12**(7), p.811.

Popova, E.D., Ahmed, A.A., Genov, P.V., Hodzha, M.M. and Zlatanova, D.P., 2019. Estimating population density of roe deer *Capreolus capreolus* (L.)(Artiodactyla: Cervidae) for better management: comparison of three methods. *Acta Zoologica Bulgarica*, **71**(4), pp.531-536.

Posit team 2024. RStudio: Integrated Development Environment for R. Posit Software, PBC, Boston, MA. URL http://www.posit.co/.

Power, M.E., Tilman, D., Estes, J.A., Menge, B.A., Bond, W.J., Mills, L.S., Daily, G., Castilla, J.C., Lubchenco, J. and Paine, R.T., 1996. Challenges in the quest for keystones: identifying keystone species is difficult—but essential to understanding how loss of species will affect ecosystems. *BioScience*, **46**(8), pp.609-620.

Prentice, J.C., Fox, N.J., Hutchings, M.R., White, P.C., Davidson, R.S. and Marion, G., 2019. When to kill a cull: factors affecting the success of culling wildlife for disease control. *Journal of the Royal Society Interface*, **16**(152), p.20180901.

Price, M.V., Strombom, E.H. and Blumstein, D.T., 2014. Human activity affects the perception of risk by mule deer. *Current Zoology*, **60**(6), pp.693-699.

Purvis, A., Gittleman, J.L., Cowlishaw, G. and Mace, G.M., 2000. Predicting extinction risk in declining species. *Proceedings of the royal society of London. Series B: Biological Sciences*, **267**(1456), pp.1947-1952.

Putman, R., 1984. Facts from faeces. *Mammal review*, **14**(2), pp.79-97.

Putman, R. 1987. The Pressure of Grazing and its Impact Upon Vegetation. In: *Grazing in Temperate Ecosystems: Large Herbivores and the Ecology of the New Forest*. Beckenham: Croom Helm. pp. 134-164.

Putman, R., 2010. Ungulates and their management in Great Britain and Ireland. In Apollonio, M., Andersen, E. and Putman, R. eds. *European ungulates and their management in the 21st century*. New York: Cambridge University Press. pp.129-64.

Putman, R., 2011. A review of the various legal and administrative systems governing management of large herbivores in Europe. In: Putman, R.J., Apollonio, M. and Andersen, R. eds. *Ungulate Management in Europe: Problems and Practices*. Cambridge: Cambridge University Press, pp.54-79.

Putman, R. 2024. *British Deer and Their Management. A Personal Reflection*. Great Britain: Amazon

Putman, R. and Apollonio, M. 2014. *Behaviour and management of European ungulates*. Dunbeath, UK: Whittles Publishing.

Putman, R.J. and Moore, N.P. 1998. Impact of deer in lowland Britain on agriculture, forestry and conservation habitats. *Mammal Review*, **28**(4), pp.141-164.

Putman, R., Langbein, J. and Staines, B.W., 2004. *Deer and Road Traffic Accidents, a Review of Mitigation Measures: Costs and Cost-effectiveness*. Inverness, UK: Scottish Natural Heritage.

Putman, R., Langbein, J., Green, P. and Watson, P., 2011a. Identifying threshold densities for wild deer in the UK above which negative impacts may occur. *Mammal Review*, **41**(3), pp. 175-196.

Putman, R., Watson, P. and Langbein, J., 2011b. Assessing deer densities and impacts at the appropriate level for management: a review of methodologies for use beyond the site scale. *Mammal Review*, **41**(3), pp. 197-219.

Putman, R.J., Langbein, J., Watson, P., Green, P. and Cahill, S., 2014. The management of urban populations of ungulates. *In:* Putman, R. and Apollonio, M. (eds). *Behaviour and Management of European Ungulates*, Dunbeath: Whittles Publishing, pp.148-177.

Putman, R., Krojerová-Prokešová. J., Barančeková. M., 2022. Sika *Cervus nippon* (Temminck, 1838). In: Corlatti, L. and Zachos, F.E. eds. *Terrestrial Cetartiodactyla, Handbook of the Mammals of Europe*. Cham: Springer Nature. pp. 87-114.

R Core Team 2024. R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. URL https://www.R-project.org/.

Rademaker, M., Meijaard, E., Semiadi, G., Blokland, S., Neilson, E.W. and Rode-Margono, E.J., 2016. First ecological study of the Bawean warty pig (*Sus blouchi*), one of the rarest pigs on earth. *Plos One*, **11**(4), p.e0151732.

Rahman, D.A., Gonzalez, G. and Aulagnier, S., 2017. Population size, distribution and status of the remote and Critically Endangered Bawean deer Axis kuhlii. *Oryx*, **51**(4), pp.665-672.

Ramirez, J.I., 2021. Uncovering the different scales in deer–forest interactions. *Ecology and Evolution*, **11**(10), pp.5017-5024.

Ramón-Laca, A., Gleeson, D., Yockney, I., Perry, M., Nugent, G. and Forsyth, D.M., 2014. Reliable discrimination of 10 ungulate species using high resolution melting analysis of faecal DNA. *Plos one*, **9**(3), p.e92043.

Ramos, S. 2022. Understanding Yurok traditional ecological knowledge and wildlife management. *The Journal of Wildlife Management*. **86**(1), p.e22140.

Ramirez, J.I., 2021. Uncovering the different scales in deer–forest interactions. *Ecology and Evolution*, **11**(10), pp. 5017-5024.

Ramos, S. 2022. Understanding Yurok traditional ecological knowledge and wildlife management. *The Journal of Wildlife Management*. **86**(1), p.e22140.

Rao, S.J., 2017. Effect of reducing red deer *Cervus elaphus* density on browsing impact and growth of Scots pine Pinus sylvestris seedlings in semi-natural woodland in the Cairngorms, UK. *Conservation Evidence*, **14**, p.22.

Rapport, D.J., Gaudet, C., Karr, J.R., Baron, J.S., Bohlen, C., Jackson, W., Jones, B., Naiman, R.J., Norton, B. and Pollock, M.M., 1998. Evaluating landscape health: integrating societal goals and biophysical process. *Journal of Environmental Management*, **53**(1), pp.1-15.

Raymond, C.M., Bieling, C., Fagerholm, N., Martin-Lopez, B. and Plieninger, T., 2016. The farmer as a landscape steward: Comparing local understandings of landscape stewardship, landscape values, and land management actions. *Ambio*, **45**, pp.173-184.

Rea, A.W. and Munns Jr, W.R., 2017. The value of nature: Economic, intrinsic, or both?. *Integrated Environmental Assessment and Management*, **13**(5), pp.953-955.

Redpath, S.M., Young, J., Evely, A., Adams, W.M., Sutherland, W.J., Whitehouse, A., Amar, A., Lambert, R.A., Linnell, J.D., Watt, A. and Gutierrez, R.J. 2013. Understanding and managing conservation conflicts. *Trends in ecology & evolution*, **28**(2), pp.100-109.

Reed, S.P., Royo, A.A., Fotis, A.T., Knight, K.S., Flower, C.E. and Curtis, P.S., 2022. The long-term impacts of deer herbivory in determining temperate forest stand and canopy structural complexity. *Journal of Applied Ecology*, **59**(3), pp. 812-821.

Reimoser, F. and Gossow, H., 1996. Impact of ungulates on forest vegetation and its dependence on the silvicultural system. *Forest Ecology and Management*. **88**, pp. 107-119.

Reimoser, F. and Nopp-Mayr, U., 2024. Ecology, diversity, conservation and management of ungulates. *Diversity*, **16**(3), p.182.

Reimoser, F. and Putman, R., 2011. Impact of wild ungulates on vegetation: costs and benefits. In: Putman, R.J., Apollonio, M. and Andersen, R. eds. *Ungulate Management* *in Europe: Problems and Practices***.** Cambridge: Cambridge University Press. pp. 144-191.

Richardson, S., Mill, A.C., Davis, D., Jam, D. and Ward, A.I. 2020. A systematic review of adaptive wildlife management for the control of invasive, non‐native mammals, and other human‐wildlife conflicts. *Mammal Review*. **50**(2), pp.147-156.

Riechers, M., Balázsi, Á., Betz, L., Jiren, T.S. and Fischer, J., 2020. The erosion of relational values resulting from landscape simplification. *Landscape Ecology*, **35**, pp.2601-2612.

Riesch, F., Tonn, B., Stroh, H.G., Meißner, M., Balkenhol, N. and Isselstein, J., 2020. Grazing by wild red deer maintains characteristic vegetation of semi-natural open habitats: Evidence from a three-year exclusion experiment. *Applied Vegetation Science*, **23**(4), pp.522-538.

Riley, S.J., Decker, D.J., Carpenter, L.H., Organ, J.F., Siemer, W.F., Mattfeld, G.F. and Parsons, G., 2002. The essence of wildlife management. *Wildlife Society Bulletin*, **30**(2), pp.585-593.

Riley, S.J., Siemer, W., Decker, D.J., Carpenter, L., Organ, J.F. and Berchielli, L., 2003. Adaptive impact management: an integrative approach to wildlife management. *Human Dimensions of Wildlife*, *8*(2), pp.81-95.

Robin, L., 2017. Culling and care: ferals, invasives and conservation icons in Australia. *Australian Zoologist*, **39**(1), pp.103-113.

Robinson, K.F., Fuller, A.K., Hurst, J.E., Swift, B.L., Kirsch, A., Farquhar, J., Decker, D.J. and Siemer, W.F., 2016. Structured decision making as a framework for large-scale wildlife harvest management decisions. *Ecosphere*, **7**(12), p.e01613.

Rowcliffe, J.M., Field, J., Turvey, S.T. and Carbone, C., 2008. Estimating animal density using camera traps without the need for individual recognition. *Journal of Applied Ecology,* **45**(4), pp. 1228-1236

Rowcliffe, J.M., Jansen, P.A., Kays, R., Kranstauber, B. and Carbone, C., 2016. Wildlife speed cameras: measuring animal travel speed and day range using camera traps. *Remote Sensing in Ecology and Conservation*, **2**(2), pp.84-94.

Royo, A.A., Stout, S.L., deCalesta, D.S. and Pierson, T.G., 2010. Restoring forest herb communities through landscape-level deer herd reductions: Is recovery limited by legacy effects?. *Biological Conservation*, **143**(11), pp. 2425-2434.

Ruskule, A., Nikodemus, O., Kasparinskis, R., Bell, S. and Urtane, I., 2013. The perception of abandoned farmland by local people and experts: Landscape value and perspectives on future land use. *Landscape and Urban Planning*, **115**, pp.49-61.

Rust, N.A. 2017. Can stakeholders agree on how to reduce human–carnivore conflict on Namibian livestock farms? A novel Q-methodology and Delphi exercise. *Oryx*. **51**(2), pp.339-346.

Sakurai, R., Uehara, T., Tsunoda, H., Enari, H., Stedman, R.C. and Onuma, A., 2024. Developing a system model for articulating the social-ecological impacts of species reintroduction. *Ecology and Society*, **29**(2).

Salerno, J., Aguirre, A.A., Aguirre, L.F., Bonacic, C., Bosco-Lauth, A., Breck, S.W., Gitonga, P.N., Goldstein, T., Gonzalez, M.N., Hoag, D.L.K. and Mayo, C., 2025. Transdisciplinary research priorities for a One Health approach to human–wildlife coexistence. *BioScience*, p.biaf026. https://doi.org/10.1093/biosci/biaf026

Saldaña, J. 2021. *The coding manual for qualitative researchers*. 2nd ed. London: SAGE.

Salles, J.M., 2011. Valuing biodiversity and ecosystem services: Why put economic values on Nature?. *Comptes Rendus. Biologies*, **334**(5-6), pp.469-482.

Sánchez-Rojas, G., Gallina, S. and Equihua, M., 2004. Pellet morphometry as a tool to distinguish age and sex in the mule deer. *Zoo Biology: Published in Affiliation with the American Zoo and Aquarium Association*, **23**(2), pp.139-146.

Salvati, L., Tombolini, I., Perini, L. and Ferrara, A., 2013. Landscape changes and environmental quality: the evolution of land vulnerability and potential resilience to degradation in Italy. *Regional Environmental Change*, **13**(6), pp.1223-1233.

Salvatori, V., Balian, E., Blanco, J.C., Ciucci, P., Demeter, L., Hartel, T., Marsden, K., Redpath, S.M., von Korff, Y. and Young, J.C. 2020. Applying Participatory Processes to Address Conflicts Over the Conservation of Large Carnivores: Understanding Conditions for Successful Management. *Frontiers in Ecology and Evolution*. **8**, p.182.

Salvatori, V., Balian, E., Blanco, J.C., Carbonell, X., Ciucci, P., Demeter, L., Marino, A., Panzavolta, A., Sólyom, A., von Korff, Y. and Young, J.C. 2021. Are large carnivores the real issue? Solutions for improving conflict management through stakeholder participation. *Sustainability*. **13**(8), p.4482.

Sand, H., Jamieson, M., Andrén, H., Wikenros, C., Cromsigt, J. and Månsson, J., 2021. Behavioral effects of wolf presence on moose habitat selection: testing the landscape of fear hypothesis in an anthropogenic landscape. *Oecologia*, **197**, pp.101-116.

Sanders, D. and Frago, E., 2024. Ecosystem engineers shape ecological network structure and stability: A framework and literature review. *Functional Ecology*, **38**(8), pp.1683-1696.

Santoro, A., 2024. Why traditional rural landscapes are still important to our future. *Landscape Ecology*, **39**(8), p.135.

Schippers, P., van der Heide, C.M., Koelewijn, H.P., Schouten, M.A., Smulders, R.M., Cobben, M.M., Sterk, M., Vos, C.C. and Verboom, J., 2015. Landscape diversity enhances the resilience of populations, ecosystems and local economy in rural areas. *Landscape Ecology*, **30**(2), pp.193-202.

Scottish Natural Heritage, 2016. *Deer management in Scotland: report to the Scottish Government from Scottish Natural Heritage*, Inverness: s.n.

Segar, J., Pereira, H.M., Filgueiras, R., Karamanlidis, A.A., Saavedra, D. and Fernández, N. 2022. Expert-based assessment of rewilding indicates progress at site-level, yet challenges for upscaling. *Ecography*. **2022**(4), pp.1-10.

Servanty, S., Gaillard, J.M., Ronchi, F., Focardi, S., Baubet, E. and Gimenez, O., 2011. Influence of harvesting pressure on demographic tactics: implications for wildlife management. *Journal of Applied Ecology*, **48**(4), pp.835-843.

Siemer, W.F., Lauber, T.B., Chase, L.C., Decker, D.J., McPeake, R.J. and Jacobson, C.A. 2004. Deer/elk management actions in suburban environments: what will stakeholders accept? *Proceedings of the 4th International Urban Wildlife Symposium on Urban Wildlife Conservation, 2004, Tucson, Arizona, USA*. pp.228-237. Available from: https://cales.arizona.edu/pubs/adjunct/snr0704/snr07042t.pdf

Silovský, V., Landler, L., Faltusová, M., Börger, L., Burda, H., Holton, M., Lagner, O., Malkemper, E.P., Olejarz, A., Spießberger, M. and Váchal, A., 2024. A GPS assisted translocation experiment to study the homing behavior of red deer. *Scientific Reports*, **14**(1), p.6770.

Sinclair, A.R. 1991. Science and the practice of wildlife management. *The Journal of wildlife management*. **55**(4), pp.767-773.

Smart, J.C., Ward, A.I. and White, P.C., 2004. Monitoring woodland deer populations in the UK: an imprecise science. *Mammal Review*, **34**(1-2), pp.99-114.

Smit, C. and Putman, R.J., 2011. Large herbivores as ‘environmental engineers’. *In:* Putman, R.J., Apollonio, M. and Andersen, R. (eds). *Ungulate Management in Europe: Problems and Practices*, Cambridge: Cambridge University Press, pp.260-283.

Smith, B.P., Appleby, R.G. and Jordan, N.R. 2021. Co-existing with dingoes: Challenges and solutions to implementing non-lethal management. *Australian Zoologist*. **41**(3), pp.491-510.

Smith, E., 2025. *Spatial and behavioural impacts of disturbance on red deer, Cervus elaphus, in the Scottish uplands.* PhD thesis, Durham University.

Soga, M. and Gaston, K.J., 2016. Extinction of experience: the loss of human–nature interactions. *Frontiers in Ecology and the Environment*, **14**(2), pp.94-101.

Songhurst, A., 2017. Measuring human–wildlife conflicts: Comparing insights from different monitoring approaches. *Wildlife Society Bulletin*, **41**(2), pp.351-361.

Soofi, M., Ghoddousi, A., Hamidi, A.K., Ghasemi, B., Egli, L., Voinopol-Sassu, A.J., Kiabi, B.H., Balkenhol, N., Khorozyan, I. and Waltert, M., 2017. Precision and reliability of indirect population assessments for the Caspian red deer Cervus elaphus maral. *Wildlife Biology*, **2017**(1), pp.1-8.

Spake, R., Bellamy, C., Gill, R., Watts, K., Wilson, T., Ditchburn, B. and Eigenbrod, F., 2020. Forest damage by deer depends on cross-scale interactions between climate, deer density and landscape structure. *Journal of Applied Ecology*, **57**(7), pp.1376-1390.

Spitzer, R., Churski, M., Felton, A., Heurich, M., Kuijper, D.P., Landman, M., Rodriguez, E., Singh, N.J., Taberlet, P., van Beeck Calkoen, S.T. and Widemo, F.,

2019. Doubting dung: eDNA reveals high rates of misidentification in diverse European ungulate communities. *European Journal of Wildlife Research*, **65**(2), p.28.

Staines, B.W. and Welch, D., 1984. Habitat selection and impact of red (Cervus elaphus L.) and roe (*Capreolus capreolus L.*) deer in a Sitka spruce plantation. *Proceedings of the Royal Society of Edinburgh, Section B: Biological Sciences*, **82**(4), pp.303-319.

Steiner, W., Schöll, E.M., Leisch, F. and Hackländer, K., 2021. Temporal patterns of roe deer traffic accidents: Effects of season, daytime and lunar phase. *PLoS One*, *16*(3), p.e0249082.

Stokes, M., 2011. An introduction to managing wild game populations. *Nature Education Knowledge*, **2**(11), p.4.

Swan, G.J., Redpath, S.M., Bearhop, S. and McDonald, R.A., 2017. Ecology of problem individuals and the efficacy of selective wildlife management. *Trends in Ecology & Evolution*, **32**(7), pp.518-530.

Sykes, N. (2004). The Introduction of Fallow Deer to Britain: A Zooarchaeological Perspective. *Environmental Archaeology*, **9**(1), pp.75–83.

Tabor, R.C., 1993. Control of deer in a managed coppice. *Quarterly Journal of Forestry* **87**(4), pp.308-313.

Tanentzap, A.J., Bazely, D.R., Koh, S., Timciska, M., Haggith, E.G., Carleton, T.J. and Coomes, D.A., 2011. Seeing the forest for the deer: do reductions in deer-disturbance lead to forest recovery?. *Biological Conservation*, **144**(1), pp.376-382.

Tanentzap, A.J., Kirby, K.J. and Goldberg, E., 2012. Slow responses of ecosystems to reductions in deer (Cervidae) populations and strategies for achieving recovery. *Forest Ecology and Management*, **264**, pp.159-166.

Taylor, K., 2019. New lives, new landscapes. Landscape, heritage and rural revitalisation: Whose cultural values?. *Built Heritage*, **3**(2), pp.50-63.

Thirgood, S.J., Redpath, S.M., Haydon, D.T., Rothery, P., Newton, I. and Hudson, P.J., 2000. Habitat loss and raptor predation: disentangling long–and short–term causes of red grouse declines. *Proceedings of the Royal Society of London. Series B: Biological Sciences*, **267**(1444), pp.651-656.

Thomas, L., Buckland, S.T., Rexstad, E.A., Laake, J.L., Strindberg, S., Hedley, S.L., Bishop, J.R., Marques, T.A. and Burnham, K.P., 2010. Distance software: design and analysis of distance sampling surveys for estimating population size. *Journal of Applied Ecology*, **47**(1), pp.5-14.

Thompson, D.B.A., MacDonald, A.J., Marsden, J.H. and Galbraith, C.A., 1995. Upland heather moorland in Great Britain: a review of international importance, vegetation change and some objectives for nature conservation. *Biological Conservation*, **71**(2), pp.163-178.

Tixier, H. and Duncan, P., 1996. Are European roe deer browsers? A review of variations in the composition of their diets. *Revue D'écologie*, **51**(1), pp.3-17.

Tixier, H., Duncan, P., Scehovic, J., Yant, A., Gleizes, M. and Lila, M., 1997. Food selection by European roe deer (Capreolus capreolus): effects of plant chemistry, and consequences for the nutritional value of their diets. *Journal of Zoology*, **242**(2), pp.229-245.

Treves, A., Wallace, R.B., Naughton-Treves, L. and Morales, A., 2006. Co-managing human–wildlife conflicts: a review. *Human dimensions of wildlife*, **11**(6), pp.383-396.

Tribot, A.S., Deter, J. and Mouquet, N., 2018. Integrating the aesthetic value of landscapes and biological diversity. *Proceedings of the Royal Society B: Biological Sciences*, **285**(1886), p.20180971.

Ueno, M., Kaji, K. and Saitoh, T., 2010. Culling versus density effects in management of a deer population. *The Journal of Wildlife Management*, **74**(7), pp.1472-1483.

UNESCO. 2025. The English Lake District. [Online]. [Accessed 9 August 2025]. Available from: https://whc.unesco.org/en/list/422/

Urbanek, R.E., Nielsen, C.K., Davenport, M.A. and Woodson, B.D., 2013. Determinants of public perceptions of suburban deer density. *Human Dimensions of Wildlife*, **18**(2), pp.82-96.

Valente, A.M., Acevedo, P., Figueiredo, A.M., Fonseca, C. and Torres, R.T., 2020a. Overabundant wild ungulate populations in Europe: management with consideration of socio-ecological consequences. *Mammal Review*, **50**(4), pp.353-366.

Valente, A.M., Acevedo, P., Figueiredo, A.M., Martins, R., Fonseca, C., Torres, R.T. and Delibes-Mateos, M., 2020b. Dear deer? Maybe for now. People's perception on red deer (Cervus elaphus) populations in Portugal. *Science of the Total Environment*, **748**, p.141400.

van Beeck Calkoen, S.T., Kuijper, D.P., Apollonio, M., Blondel, L., Dormann, C.F., Storch, I. and Heurich, M., 2023. Numerical top-down effects on red deer (Cervus elaphus) are mainly shaped by humans rather than large carnivores across Europe. *Journal of Applied Ecology*, **60**(12), pp.2625-2635.

Van der Elst, J., Rissetto-Richards, H., and Diaz-Kommonen, L., 2018. Rural Sense: Value, Heritage, and Sensory Landscapes: Developing a design-directed approach to mapping for healthier landscapes. *Landscape Review*, **18**(1), pp.56-71

van de Water, A., Doornwaard, S., Sluiter, L., Henley, M., Sutherland, C. and Slotow, R. 2023. Resolving conservation conflicts through shared vision, collective benefits and relevant values. *Diversity*, **15**(10), p.1041.

van Poorten, B. and Beck, M. 2021. Getting to a decision: using structured decision-making to gain consensus on approaches to invasive species control. *Management of Biological Invasions*. **12**(1), p.25.

Vasudeva, V., Ramasamy, P., Pal, R.S., Behera, G., Karat, P.R. and Krishnamurthy, R., 2021. Factors influencing people's response toward tiger translocation in Satkosia Tiger Reserve, Eastern India. *Frontiers in Conservation Science*, **2**, p.664897.

Vatn, A., Pascual, U., Chaplin-Kramer, R., Termansen, M., Arias-Arévalo, P., Balvanera, P., Athayde, S., Hahn, T. and Lazos, E., 2024. Incorporating diverse values of nature in decision-making—theory and practice. *Philosophical Transactions of the Royal Society B*, **379**(1903), p.20220315.

Venables, W.N. and Ripley, B.D. 2002. *Modern Applied Statistics with S*. 4th Edition. New York: Springer.

Vindevoghel, V., 2024. Rethinking the geography of sustainability transitions by considering human-nature connections in rural areas. *Environmental Innovation and Societal Transitions*, **51**, p.100851.

Viollaz, J.S., Thompson, S.T. and Petrossian, G.A., 2021. When human–wildlife conflict turns deadly: Comparing the situational factors that drive retaliatory leopard killings in South Africa. *Animals*, **11**(11), p.3281.

Von Essen, E., 2018. The impact of modernization on hunting ethics: Emerging taboos among contemporary Swedish hunters. *Human Dimensions of Wildlife*, **23**(1), pp.21-38.

Von Essen, E. and Redmalm, D., 2023. Social licence to cull: Examining scepticism toward lethal wildlife removal in cities. *People and Nature*, **5**(4), pp.1353-1363.

Von Hagen, L.Y.N.N., Gray, S.A., Schulte, B.A., Githiru, M., Kiute, H.I. and Lepczyk, C.A. 2024. Participatory modelling across Kenyan villages facilitates insights into the complexity of human–elephant interactions. *Oryx*, pp.1-10.

Walter, W.D., Lavelle, M.J., Fischer, J.W., Johnson, T.L., Hygnstrom, S.E. and VerCauteren, K.C., 2010. Management of damage by elk (Cervus elaphus) in North America: a review. *Wildlife Research*, **37**(8), pp.630-646.

Ward, A.I., 2001. *The ecology and sustainable management of roe deer (Capreolus capreolus) in multiple-use forestry*. PhD thesis, University of York.

Ward, A.I., 2005. Expanding ranges of wild and feral deer in Great Britain. *Mammal Review,* **35**(2), pp. 165-173

Ward, A.I., 2007. *Trends in deer distribution and abundance within the UK* [Presentation]. Deer Initiative Conference: Deer, Habitats and Impacts, 23 March 2007. Buxton.

Ward, A.I., Richardson, S., Macarthur, R. and Mill, A.C., 2020. Using and communicating uncertainty for the effective control of invasive non-native species. *Mammal review*, **50**(2), pp.211-220.

Ward, A.I., White, P.C. and Critchley, C.H., 2004. Roe deer *Capreolus capreolus* behaviour affects density estimates from distance sampling surveys. *Mammal Review*, **34**(4), pp.315-319.

Ward, A.I., White, P.C., Smith, A. and Critchley, C.H., 2004. Modelling the cost of roe deer browsing damage to forestry. *Forest Ecology and Management*, **191**(1-3), pp.301-310.

Ward, A.I., White, P.C.L., Walker, N.J., Critchley, C.H., 2008. Conifer leader browsing by roe deer in English upland forests: Effects of deer density and understorey vegetation. *Forest Ecology and Management.* **256**(6), pp. 1333–1338.

Ward, J.S. and Williams, S.C., 2010. Effectiveness of deer repellents in Connecticut. *Human-Wildlife Interactions*, **4**(1), pp.56-66.

Wardropper, C. B , Graves, R.A., Brandt, J., Burnham, M., Carter, N., Hale, R.L., Hillis, V. and Williamson, M.A., 2024. Private land conservation towards large landscape goals: Role of relational values, property rights orientations and perceived efficacy in ranchers' actions. *People and Nature*, **6**(3), pp. 1171-1188

Waters, W.R. and Palmer, M.V., 2015. Mycobacterium bovis infection of cattle and white-tailed deer: translational research of relevance to human tuberculosis. *ILAR journal*, **56**(1), pp.26-43.

Welch, D., Staines, B.W., Scott, D. and Catt, D.C., 1987. Bark stripping damage by red deer in a Sitka spruce forest in Western Scotland. I. Incidence. *Forestry: An International Journal of Forest Research*, **60**(2), pp.249-262.

White, P.C. and Ward, A.I. 2010. Interdisciplinary approaches for the management of existing and emerging human–wildlife conflicts. *Wildlife Research*, **37**(8), pp.623-629.

White, R.M., Schmook, B., Calmé, S., Giordano, A.J., Hausser, Y., Kimmel, L., Lecuyer, L., Lucherini, M., Méndez-Medina, C. and Peña-Mondragón, J.L., 2023. Facilitating biodiversity conservation through partnerships to achieve transformative outcomes. *Conservation Biology*, **37**(3), p.e14057.

Whitefield, A.C., McMorran, R., Paterson, J.S. and Warren, C.R., 2021. Public perceptions of deer management in Scotland: the impact of place of residence, knowledge and demographic factors. *Scottish Geographical Journal*, **137**(1-4), pp.67-83.

Whitehead, T. and Hare, D., 2025. A shifting baseline theory of debates over potential lynx and wolf reintroductions to Scotland. *Ambio*, pp.1-13.

Wild Deer Best Practice Guidance for Scotland, 2018. *Habitat Impacts*. Available from: https://bestpracticeguides.org.uk/impacts/

Wild Deer Best Practice Guidance for Scotland, 2026. *Planning*. Available from: https://bestpracticeguides.scot/wp-content/uploads/2026/01/2-Planning-Drones-2.pdf

Wilkie, D.S., Wieland, M. and Poulsen, J.R., 2019. Unsustainable vs. sustainable hunting for food in Gabon: Modeling short-and long-term gains and losses. *Frontiers in Ecology and Evolution*, **7**, p.357.

Williams, B.K. 2011a. Adaptive management of natural resources—framework and issues. *Journal of Environmental Management*. **92**(5), pp.1346-1353.

Williams, B.K. 2011b. Passive and active adaptive management: approaches and an example. *Journal of Environmental Management*. **92**(5), pp.1371-1378.

Williams, B.K. and Brown, E.D. 2014. Adaptive management: from more talk to real action. *Environmental Management*. **53**(2), pp.465-479.

Williams, B.K. and Brown, E.D. 2016. Technical challenges in the application of adaptive management. *Biological Conservation*. **195**, pp.255-263.

Williams, B.K., Eaton, M.J. and Breininger, D.R., 2011. Adaptive resource management and the value of information. *Ecological Modelling*, **222**(18), pp.3429-3436.

Wilson, S. and Campera, M., 2024. The Perspectives of Key Stakeholders on the Reintroduction of Apex Predators to the United Kingdom. *Ecologies*, **5**(1), pp.52-67.

Wohlfahrt, S., Balkenhol, N., Edelhoff, H., Anderwald, P., Signer, J., Leitner, H., Klück, P., Schueler, S., Langmaier, M., Kohlross, H. and Hackländer, K., 2025. Exploring the use of REM in compact hunting grounds comparing site specific to average parameters. *European Journal of Wildlife Research*, **71**(2), p.34.

Wójcicki, A. and Borowski, Z., 2023. The presence of wolves leads to spatial differentiation in deer browsing pressure on forest regeneration. *Scientific Reports*, **13**(1), p.17245.

Woolaston, K., Flower, E., Van Velden, J., White, S., Burns, G.L. and Morrison, C. 2021. A review of the role of law and policy in human-wildlife conflict. *Conservation and Society*. **19**(3), pp.172-183.

Xie, H., Zhu, Z., He, Y., Zeng, X. and Wen, Y., 2022. Integrated framework of rural landscape research: Based on the global perspective. *Landscape Ecology*, **37**(4), pp.1161-1184

Yeshey, Keenan, R.J., Ford, R.M. and Nitschke, C.R., 2024. Social and ecological dimensions are needed to understand human-wildlife conflict in subsistence farming context. *People and Nature*, **6**(6), pp.2602-2617.

Yousuf, M.I. 2019. Using experts opinions through Delphi technique. *Practical Assessment, Research, and Evaluation*. **12**(1), p.4.

Yuan, B.D., Xie, S.B., Liu, B., Xue, D.D. and Sun, D.M., 2019. Differential movement pattern of Père David's deer associated with the temporal rhythm using GPS collar fix. *Global Ecology and Conservation*, **18**, p.e00641.

Yu, Z., Nukina, R., Xie, Y. and Shibata, S., 2024. Public attitudes to urban wild deer (Cervus nippon) and management policies: A case study of Kyoto City, Japan. *Global Ecology and Conservation*, **51**, p.e02927.

Zimmermann, A., McQuinn, B. and Macdonald, D.W. 2020. Levels of conflict over wildlife: Understanding and addressing the right problem. *Conservation Science and Practice*, **2**(10), p.e259.

Zimmermann, A. and Stevens, J., 2021. Call for holistic, interdisciplinary and multilateral management of human–wildlife conflict and coexistence. *Oryx*, **55**(4), pp.490-491.

Appendix A. Ethical Approvals

Dear Thomas

BIOSCI 22-002 - Reconciling lethal control of an iconic native species (red deer Cervus elaphus) with nature conservation in a multiple-use landscape

NB: All approvals/comments are subject to compliance with current University of Leeds and UK Government advice regarding the Covid-19 pandemic.

I am pleased to inform you that the above research study has undergone ethical scrutiny by the Faculty of Biological Sciences Research Ethics Committee (FBS FREC) and on behalf of the Chair, I can confirm a favourable ethical opinion based on the documentation received at date of this email.

Please retain this email as evidence of approval in your study file.

Please note this will also need approval from Faculty of Biological Sciences Health and Safety.

Please notify the committee if you intend to make any amendments to the original research as submitted and approved to date. This includes recruitment methodology; all changes must receive ethical approval prior to implementation. Please see https://ris.leeds.ac.uk/research-ethics-and-integrity/applying-for-an-amendment/ or contact the Research Ethics Administrator for further information (FBSResearchEthics@leeds.ac.uk) if required.

Confirmation of ethical scrutiny does not infer you have the right of access to any member of staff or student or documents and the premises of the University of Leeds. Nor does it imply any right of access to the premises of any other organisation, including clinical areas. The committee takes no responsibility for you gaining access to staff, students and/or premises prior to, during or following your research activities.

Please note: You are expected to keep a record of all your approved documentation, as well as documents such as sample consent forms, risk assessments and other documents relating to the study. This should be kept in your study file, which should be readily available for audit purposes. You will be given a two week notice period if your project is to be audited.

It is our policy to remind everyone that it is your responsibility to comply with Health and Safety, Data Protection and any other legal and/or professional guidelines there may be.

I hope the study goes well.

Best wishes
Sou
On behalf of Dr David Lewis, Chair, FBS FREC

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Sou Sit Chung, Research Ethics Administrator, The Secretariat, University of Leeds, LS2 9NL, s.chung@leeds.ac.uk
Please note my working hours are Monday to Friday 9am – 12.30pm

Copy of the email from the Faculty of Biological Sciences Ethics Committee confirmation of ethical approval for the social science elements of this research.
~~~~~~

From: FBSResearchEthics <FBSResearchEthics@leeds.ac.uk>
Sent: 05 September 2022 08:41
To: Thomas Logan <eetwl@leeds.ac.uk>
Subject: BIOSCI 22-004 - Ethics not required

Hi Tom

The Chair has confirmed that ethics approval/ethical scrutiny is not required for this non-human element of the research as per the reasons below:

- This fieldwork element, surveying of animal species, is outside of the remit of our committee
- This animal work is non-regulated, purely observational, and therefore outside of the remit of the University Animal Welfare and Ethical Review Body
- The only issue for our committee is whether it poses reputational risk, as it is purely observational it doesn't.

Hope the fieldwork goes well.

Kind regards.
Sou

~~~~~~

Sou Sit Chung, Research Ethics Administrator, The Secretariat, University of Leeds, LS2 9NL, s.chung@leeds.ac.uk
Please note my working hours are Monday to Friday 9am – 12.30pm

Copy of the email from the Faculty of Biological Sciences Ethics Committee confirming that ethical approval was not required for the non-human elements of the research for the reasons outlined in the bullet points. For any research before this date, approval had been previously granted by the University of Hull's Faculty of Science and Engineering Ethics Committee: Reference FEC_2022_67.
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Appendix B. Chapter 2 and 3 Questionnaires and Briefing Notes

Delphi Round One: Briefing Materials and Questionnaire

Project Title: Reconciling lethal control of an iconic native species (red deer *Cervus elaphus*) with nature conservation in a multiple use landscape

Overview: Thank you for agreeing to participate in the first of our three questionnaires, which aims to identify the range of attitudes towards the red deer populations, and their management, amongst the stakeholders of the Borrowdale and Thirlmere regions. We aim to conduct two further questionnaires that will map out the attitudes identified in this questionnaire with data on the red deer populations. Our study will then finish by proposing some management options for the deer to key decision makers accounting for the views you have presented in this survey process.

Once the responses from all participants of this first questionnaire round have been received by the Lead Researchers, we will collate and summarise the findings for participants to reflect upon in the round two questionnaire.

Your participation in this survey process is entirely voluntary, and your personal information will be treated in confidence. Please see the participant consent form for more information on how your personal information will be stored.

For all questions in this questionnaire, the boxes can be expanded to accommodate your responses; they do not have to fit into the size of the box provided.

1. Please provide your information below.

a. Name

b. Email address

c. Organisation *please select one from the following options by placing a ☑ next to the selected response:*

National Trust		United Utilities		Forestry Commission		Natural England		Other	

Other. Please State

2. Exploring the relationship of participants to the environment.

a. What are your views on wildlife in the UK?

b. What are your views on Nature Conservation in the UK?

c. What are your views on sustainable use of the environment?

d. Do you have any experience or knowledge of human-wildlife conflict? ☑

Yes		No	

If yes, please explain your answer

3. Exploring perceptions on red deer in Borrowdale and Thirlmere.

a. Do you think there should be a red deer population in the Borrowdale/Thirlmere region?☑

Yes		No	

Please explain your answer

b. Do you think red deer have any impacts on the environment in Borrowdale and Thirlmere? ☑

Positive		Negative		Both		Neither	

Please explain your answer

c. Do red deer have any impacts on you: ☑

Yes		No	

If no, please go to question d. If yes, please select as many from the options below that describe how the red deer impact you.

Home Life		Leisure Activities		Work Activities		Other	

Other. Please state.

Would you describe these impacts as:

Positive		Negative		Both		Neither	

Please explain your answer

d. Do you think the current population size of the red deer in the region is: ☑

Too high		Too low		Well-balanced		Uncertain	

Please explain your answer

4. Stakeholder perceptions of wildlife and deer impact management

a. What do you understand of the term “wildlife management”?

b. Do you think we need to manage wildlife? ☑

Yes		No		Uncertain	

Please explain your answer

c. What methods do you think exist to manage wildlife?

d. What information do you think is needed to decide whether or not to manage wildlife?

e. What information do you think is needed to decide how to manage wildlife?

f. How can the wildlife management strategies you identified in question c be applied to deer?

g. In the UK, who do you think is responsible for deciding upon deer management strategies?

h. What do you know of the legislation involved in deer management?

i. What do you think the priorities should be when considering deer management strategies?

5. When considering deer impact management, what information about the deer do you consider to be important in selecting the right approach?

END OF QUESTIONNAIRE – THANK YOU FOR YOUR PARTICIPATION

Delphi Round Two: Briefing Document, and combined questionnaire and feedback document

Overview: Thank you for your participation in the first of three rounds of questionnaire surveys. We would now like to invite you to participate in the second round of this process. This round will anonymously feedback the key themes identified by participants in the first-round questionnaire to allow reflections. We will then move on to identify the information and types of data that would be required to make management decisions for the Borrowdale and Thirlmere red deer population. Upon completion of the second round, a further questionnaire will be designed to provide further reflective opportunities and identify, using the information required from stakeholders, management options for the deer population.

Once the responses from all participants of this questionnaire round have been received by the Lead Researchers, we will collate and summarise the findings for participants to reflect upon in the round three questionnaire. The round three questionnaire will focus on presenting management options with the information identified in this round.

Our study will then finish by presenting deer management options to key decision makers, accounting for the views you have presented throughout this survey process.

Your participation in this survey process is entirely voluntary, and your personal information will be treated in confidence. Please see the participant consent form for more information on how your personal information will be stored.

For all questions in this questionnaire, the boxes can be expanded to accommodate your responses; they do not have to fit into the size of the box provided.

Section 1.

This section explored the relationship of participants to the environment, conservation, sustainable practices, and human-wildlife conflicts. It is important to understand these perceptions at early stages to provide context for how management may be considered later. A summary of the responses are below:

What are your views on wildlife, conservation, and sustainability in the UK?

Nature Conservation

All participants agreed that wildlife and nature conservation is important, with national declines in wildlife and habitats showing that current actions are “failing”. It was noted by some that “localised increases” in wildlife and habitats in the UK exist, such as “habitat restoration work in the Lake District” and should be expanded. It was also noted by multiple participants that conservation requires collaboration for success.

Government Policy

Three participants suggested that Government bodies and legislation need to be able to enforce legislation to promote sustainable practices, and one participant suggested that sustainable practices should feature “in all future legislation”.

Ecosystem Services

Seven participants agreed that ecosystem services are unsustainably exploited. It is also clear from all participants that nature, conservation, and sustainable practices are all important to ensuring human uses of land, such as "tourism" and "food production". One participant said that "opportunities to enhance biodiversity should be taken" to "coexist" in the future. Another suggested that a change to "damaging practices" in farming, ecotourism, and forestry, would assist in improving sustainable practices.

Management Strategies

One participant noted that species and environments "require management" through clear evidence to enhance future populations. The enhancement of nature, conservation, and sustainability was further supported by one participant in the need to better educate people.

<u>Do you have any experience or knowledge of human-wildlife conflict?</u>

Eleven out of thirteen participants said that they had experience or knowledge of human-wildlife conflict.

Management Strategies

Two participants noted the need to improve education as they had "no knowledge" and stated that many people were "…not understanding…" of the problems caused by conflicts. Such conflicts include the impacts of wildlife and sheep, with "extensive" grazing of overlapping areas, as well as "damage to trees" and "negative impacts" to regeneration. Four participants noted the need to improve collaboration over the landscape as many conflicts are "landscape-issues" and effective actions are hindered by "hostile communities".

Evidence for management

Seven participants mentioned conflicts existing between wildlife and farmers, with three specifying the impact of deer to farming, and one focussing on "predators killing livestock" being a key conflict point. Further, five participants mentioned risk of road traffic collisions impacting wildlife communities and human safety: including "risk to life from deer hitting cars".

Ecosystem Services

Three participants mentioned the impact of ecotourism on the habitats available for wildlife, and for wildlife itself: with one participant stating "…increased tourism [leading to] declines in ground nesting birds…"

Policy

One participant mentioned the risk to wildlife from poaching, which is an illegal activity and places humans and wildlife into conflict.

Q1. Do you have any comments on the above points relating to section 1? (No word limit- please expand the text box if necessary)

Section 2.

This section explored the perceptions of participants on red deer in Borrowdale and Thirlmere. It is important to reflect upon these perceptions when considering future deer management. A summary of the responses are below:

<u>Should deer be present in Borrowdale, and do they impact the environment?</u>

All thirteen participants agreed that red deer should be in the region, with one participant saying their impacts were positive, two participants suggested impacts were negative, and the remaining 10 saying that the impacts could be either.

Nature Conservation

Deer are important to increasing to wildlife populations and habitats. One participant described that "the positives [of deer impacts] outweigh the negatives" when impacts are managed, with three participants saying they are vital to ecosystems, and seven participants saying they require protection. Four participants said that deer directly lead to woodland regeneration.

Management Strategies

All participants agree that an impact exists, with three describing the negative impacts when sheep and deer graze the same area. Further, four participants described the negative impacts to trees and "tree regeneration" when impacts are negative.

Evidence for Management

Three participants commented that the impact to farmers was negative, and two mentioned the impact to road users in the area with the risk of road traffic collisions.

Ecosystem Services

Seven participants mentioned the ecosystem service benefits of having deer positively impacting the environment. Two mentioned "woodland regeneration" as a provisioning ecosystem service and two mentioned food resources from "meat" and "venison". Four participants mentioned "stable" ecosystems which come from "native" wildlife, which six people said were enjoyable to see. Further, three participants are directly employed because of the deer

Impact of red deer upon participants

Eleven out of thirteen participants said that red deer have a direct impact upon them, with seven declared positive impact, one declared negative, and three declared both. These impacts are a combination of leisure activities and work-related activities.

<u>Population size of the red deer in Borrowdale and Thirlmere</u>

Four participants said the deer population was too high, one participant said it was too low, five participants suggested the population was well-balanced, and three participants were unsure.

Management Strategies

Seven of the participants mentioned impacts of deer being an indicator of the population, suggesting impacts to grazing, trees and property boundaries were minimal, although three participants suggested these elements were highly impacted – especially in areas where deer and sheep overlap. Two participants commented on the cooperation between neighbours ensuring effective management, and one participant suggested increasing communication would help. Four participants commented that the welfare of the herd was positive, suggesting good population size.

Evidence for management

Reductions to traffic collisions with deer was identified by three participants; suggested as why the population is balanced.

Nature Conservation

Two participants suggested that woodland regeneration is reduced by a deer population that is too high. However, four other participants suggested that the regeneration seen in woodlands suggests a balanced red deer population.

Ecosystem services

One participant said that they would “like to see more” red deer in the area, and two participants said that the deer population brought tourists to the area.

Q2. Do you have any comments on the above points relating to section 2? (No word limit- please expand the text box if necessary)

Section 3.

This section explored the perceptions and knowledge of participants on wildlife management, and the deer impact management. The understanding of wildlife management is important to generate informed plans in future deer management in Borrowdale and Thirlmere. A summary of the responses are below:

<u>What is wildlife management, is it needed, and what methods exist to manage?</u>

All thirteen participants agreed that wildlife, and deer, require management.

Management Methods

Method Theme	Participant Suggestions
Lethal Control	*<u>Ten participants</u> mentioned a variety of methods including “culling”, “firearms”, “poisoning”, and “shooting”.*
Translocations	*<u>Three participants</u> mentioned translocations by “removing” them from an area and “moving them” to another area.*
Exclusion	*<u>Six participants</u> mentioned excluding animals from an area by creating “refuge areas”, using “tree guards” and “fencing”.*
Reintroductions	*<u>One participant</u> mentioned “reintroducing” animals such as “predators” to manage wildlife.*
Breeding Control	*<u>One participant</u> mentioned restricting the breeding of wildlife by using “contraceptives” as a management method.*

Evidence for Management

Four participants said management should be determined to control the size of a population. Five participants suggested that conflicts, with one specifying “property damage”, should be considered in whether wildlife should be managed.

Policy

Three participants mentioned legislation and government policy which may help to identify if, and whether, wildlife should be managed. Two participants mentioned incentive schemes for “farming” and “forestry” to determine if management may be needed.

Management Strategies

Three participants said that education around wildlife management needs to be improved, where one suggested that “knowledge of negative impacts caused by wildlife” is limited. Two participants suggested that management should be across the landscape to improve “collaboration” and “partnership[s]”, and two participants said that management was important to improve the welfare of the population.

Nature Conservation

One participant suggested creating more national parks may assist in moving wildlife to “secure areas” where they could be better managed, and “improve the ecosystem” through collaborative practices. Three participants said that wildlife management helps halt wildlife/habitat declines, with one participant suggesting that management will aid in reversing “biodiversity decline”.

Ecosystem Services

Two participants said that management will assist in woodland regeneration, and four saying that it will provide employment, and one participant saying that “opportunities to benefit society” is provided through “nature recovery” and “stable ecosystems”.

<u>What information is needed to decide whether, and how, to manage wildlife?</u>

Evidence for Management

Decisions as to how to manage, and whether to manage, were linked by seven participants to the size of the population, one participant to the reproduction rate of the population, and two on the sex ratio, or “males to females”, of the population

Management Strategies

Nine participants said that the impacts of wildlife are linked to decisions as to whether (and how) to manage, including “damage to flora and fauna”, and “impacts to people”. Seven participants linked the importance of collaboration to strategies, including working closely with “neighbours” across “the landscape”. Three participants mentioned the need to work to “objectives” and “strategies” to manage wildlife effectively, including two references to improving “welfare” and “animal health”. Education was highlighted as important to five participants in ensuring people with “no knowledge” can support “controversial” methods, including lethal control.

<u>What methods and frameworks exist to determine and prioritise deer impact management strategies in the UK?</u>

Policy

Seven participants mentioned different government bodies involved in deer management decisions in the UK, with two mentioning the Forestry Commission, one mentioning Natural

England, one mentioning Nature Scot, and two mentioning DEFRA directly. The decision-makers were identified by six participants as being a combination of “landowners” and “licensees”, with two mentioning the involvement of the Forestry Commission in shaping decisions in England. Deer management related laws were identified by seven participants to include the Deer Act 1991. Three participants identified the provisions in the Deer Act for types of weaponry used, “cull seasons”, and existence of “night shooting” policies

Management Methods

Deer management methods have included exclusion methods (seven participants) and lethal control to “reduce populations” by shooting/culling (eight participants). Exclusions were suggested to include tree guards (three participants), removal of deer (one participant) and fencing (three participants).

Q3. Do you have any comments on the above points made in Section 3? (No word limit- please expand the text box if necessary)

Section 4

Information required, and data method desired, to make deer impact management decisions in Borrowdale and Thirlmere.

<u>What information do you consider important for selecting deer impact management approaches?</u>

The table below summarises the final question of the first-round questionnaire. Under the right-side column, please rank the priority of the themes from 1 (highest priority) to 7 (lowest priority) for information you would want to make deer impact management decisions.

Theme	Information requested	Ranked Importance
Population Size	*Ten participants wished to understand the “number of deer” and the “density of deer” to be “confident of the population size”.*	… / 7
Range and Movement	*Eight participants wanted to understand “where they are” and the “range and movement” habits of the red deer.*	… / 7
Population Dynamics	*Four participants wanted to know the “male to female” ratio of the red deer population, and two participants wanted to know the reproductive rate.*	… / 7
Animal Welfare	*Five participants were concerned about the “health” and “welfare” of the population.*	… / 7

Deer Impacts	*Ten participants wanted to know about the impacts of deer to "tree regeneration" and "grazing" damage to SSSIs and the open fells.*	… / 7
Traffic collisions	*Four participants wanted information on the rates of "RTCs" and "RTAs" involving deer in the local area.*	… / 7
Collaborations	*Three participants wanted information on how partners, such as "neighbours", would "collaborate" to manage deer across the "landscape"*	… / 7

Q4. Please add any further comments you may have about your priorities information themes above for selecting management decisions.

Q5. What do you think the priority objectives are/should be for the long-term management of the Open-Hill and Woodland environments across the Borrowdale and Thirlmere Landscape

Q6. Given your answer to Q5, how do red deer feature within your stated priority objectives for long-term management of the Open-Hill and Woodland environments across the Borrowdale and Thirlmere Landscape?

Q7. Given your answers to previous two questions, do you think your priority list for information from the previous table would now change?

Yes		No		Uncertain	

Please explain your answer – if yes, please write out the new order of the themes from highest priority to lowest

<u>What type of data about the red deer in Borrowdale do you consider important for selecting impact management approaches?</u>

For the information required to decide deer impact management strategies, certain elements (such as population size, range and movement, and deer impact) can be gathered and explained by using complex statistical methods, or by more simplistic industry standard methods. Both statistical methods and industry standard methods have their merits and their drawbacks (such as cost, accuracy, training, time etc…), some of which are summarised below.

<u>Statistical Methods</u>: statistical methods require clear and consistent data collection protocols with complex analytical skills to provide accurate and reliable answers to questions asked.

However, they require extensive training to both complete and analyse, are generally costly to enact and can take a long time to collect the amount of data required to increase confidence in the analyses. Such methods can apply to population size estimates (as an upper and lower population size ranges) and deer impact assessments (fine-scale measurements providing precise impact values) that stand up to academic rigour.

<u>Industry Standard Methods</u>: industry standard methods also require a clear and consistent data collection protocol, but less complexity. This ensures that only minimal training is required to collect and interpret the data and at lower costs than statistical methods would usually incur, but the confidence in such outputs is lower than statistical methods. Such methods can also apply to population size (as a 'minimum population size') and deer impacts (on a broad scale that only measures large-scale change) but only provide acceptable details of trends over time and would struggle to stand up to academic rigour.

Q8. From the following options, what type of data collection would you prefer to be performed to make informed deer impact management decisions?

Statistical Methods		Industry Standard Methods		Combination of Both	

Please explain your answer – if a combination of both, please elaborate which method type should be used for each of the themes from the table.

END OF QUESTIONNAIRE – THANK YOU FOR YOUR PARTICIPATION

Delphi Round Three: Briefing Document, and combined questionnaire and feedback document

Overview: Thank you for your participation in the first two rounds of questionnaire surveys. We would now like to invite you to participate in the final questionnaire round of this process. This round will anonymously feedback the key themes identified by participants in the second-round questionnaire and provide you with an opportunity to add your reflections. This questionnaire will then identify your preferred management options for the Borrowdale and Thirlmere red deer population.

Your participation in this survey process is entirely voluntary, and your personal information will be treated in confidence. Please see the participant consent form for more information on how your personal information will be stored.

For all questions in this questionnaire, the boxes can be expanded to accommodate your responses; they do not have to fit into the size of the box provided.

Section 1.

This section summarises the responses to feedback from the second questionnaire based on:

1. *The relationship of participants to the environment, conservation, sustainable practices, and human-wildlife conflicts.*
2. *The perceptions of participants on red deer in Borrowdale and Thirlmere.*
3. *The perceptions and knowledge of participants on wildlife management, and the deer impact management.*

A summary of the responses to feedback from this section are summarised, by each part, below.

Part 1

Almost all participants (eight) mentioned the need to improve “communication” and “education” in “conservation”, “sustainability”, and “environmental issues”. Improving education would “may well answer most of the issues raised” and would require “increased funding” to enact, raise awareness of issues, and improve landscape collaboration. This includes the issues of “poaching”, which occur at a “UK wide” scale.

Further, education may assist in improving “sustainable use” from “tourists”. Increased tourism is “important to the area” but areas are not used in a “sustainable way” and “require education to improve this”. One participant summarised that “UK based” conservation successes “involve collaboration between land managers, local people, NGO’s, and government agencies”.

Education may also improve understanding and prevention of “overgrazing” when “sheep are overstocked” and “deer are unmanaged” as they lead to a “major decline in the health of wild places”. It was acknowledged that many farmers “work to regenerate the land”, but some "agricultural practices" have livestock "stocking densities" that are too high and introduce negative "chemicals in the environment".

One participant summarised that there is "…a balanced and consistent viewpoint between all participants." But another participant noted that specific "red deer requirements are significantly more complex, [and] must be addressed before moving onto management strategies".

Part 2

All participants agreed that the deer should be present in the area, one participant stated that the "…majority of the answers seem balanced with most participants agreeing. There begins to be differing answers when perceptions are introduced (i.e., management strategies/population size), which would be swayed by personnel stakes and agendas" and it was noted by another participant that "…perceptions are all so varied". One participant suggested that the perceptions and outcomes can be improved when "collaboration and increased communication is in place between stalkers".

There was a lack of clarity on the impacts herbivores were having upon the area and was noted by a participant that the combination of deer and sheep are "decimating some woodlands in Borrowdale" requiring a clear strategy to understand and manage this.

One participant stated that in order to manage deer populations effectively, "…we need to create a vision of 'what good looks like' in terms of a sustainable red deer population for Borrowdale and Thirlmere" to "achieve a level of consensus" and ensure "the management strategy [is] aligned to long-term goals". Another participant suggested that people should "stop seeing ourselves as separate from the very ecosystem we wish to protect" to increase understanding of impacts from "wildlife and people" for clearer strategies.

Part 3

Nine participants agreed that a combination approach using multiple methods, including "culling", "exclusion", "monitoring", and "collaborations" are key to successful management. Lethal approaches are generally considered to be the "most effective". However, one participant has said that lethal approaches should only be considered if "non-lethal methods" have been "exhausted" and "should be as humane as possible and selective" to ensure "welfare and ethics are appropriately considered" in management action.

Three participants suggested that moving deer (translocations) and "contraception" are not "viable options" to "reduce populations", whilst "culling" can be used to help "prevent cruelty when [deer] are ill or injured" and is the only method that "actively reduces" populations.

Q1. Do you have any comments on the above points relating to second questionnaire feedback to Section 1? *(No word limit- please expand the text box as necessary).*

Section 2.

Information required, and data method desired, to make deer impact management decisions in Borrowdale and Thirlmere.

The importance of the data that could be used to select impact management strategies was ranked by the 11 participants in the second questionnaire. The list of data required as ranked by participants is below (1 = most important; 7 = least important):

1. *Animal Welfare,*
2. *Population Size,*
3. *Deer Impacts,*
4. *Range and Movement,*
5. *Collaborations,*
6. *Population Dynamics,*
7. *Traffic Collisions.*

Comments Feedback:

Nine respondents stated that management decisions require data on a combination of multiple factors; including herd size and dynamics, movements, and impacts – and that management should consider ensuring high welfare of the deer population.

Selected participant comments:

- “Deer welfare and impacts on the landscape should be priority”.
- “How many deer” can be “sustained” before “new growth is immediately cleared by grazing animals”.
- “Impact can be usefully linked to population size (although significant caution is required) to enable cull targets to be set”.
- There are “woodlands with deer” only and have had “no sheep grazing” for “many years” that should be an “effective [comparator] for overall impacts”.
- “What you can’t measure, you can’t manage”.

Collaboration was noted of importance by seven participants and was summarised by one participant as: “A lot has been written about ‘landscape-scale deer management’ and to effectively manage deer that move across boundaries; this really needs to happen out in the field and not just on paper”.

Q2. Do you have any comments regarding the ranking of the priority of data information themes above (and of the further comments) for management decision making? *(No word limit- please expand the text box as necessary).*

<u>What should the priority objectives be for long-term management of the Open-Hill and Woodland environments and how do deer feature in these objectives?</u>

Selected participant comments:

- “The two major landowner/managers appear to have separate objectives” and when combined should “embrace native deer and reduce [direct] competition from sheep grazing”. To enable this, “deer only zones should be designated in future agri-environment schemes and a system of compensatory payments agreed for such areas”.

- “Sustainable and healthy population of all wildlife, including red deer which should be managed”.

- “Restoration and regeneration” of “resilient habitats” and “protected sites”, and “natural flood management” to “accommodate numerous flora and fauna”. This includes “woodland merging into fell terrain and blending into mountain environment”. Further, “habitats need to be connected” through “planting, rewilding, and regenerating a range of habitats”. Deer need managing to “levels at which this can be achieved”.

- “Nature recovery” and “habitat improvement” for a “resilient landscape” of which a “sustainable herd should be able to thrive” without “negative impacts to people” by including the “impacts of all herbivores”, not just deer. This will be achieved with “improved engagement” with “landowners and farmers”, with “consideration” to “what is happening elsewhere [locally] with deer management”.

- Eight participants stated that deer should feature in “all landscape objectives” as an “important native species” that are “an important part of the ecosystem” but should be “managed” to create a “sustainable herd” that has no “negative impacts” upon people or the “landscape”.

- One participant stated that policymakers could assist landscape objectives including deer by creating “deer only zones in future agri-environment schemes and a system of compensatory payments agreed for such areas”.

- One participant commented that “roe deer should be considered” in objectives and plans moving forward, “not just the red” deer.

No participants changed their priority lists for data based upon how these interact with stated objectives.

Q3. Do you have any comments regarding the objectives guiding management and how deer feature within these objectives? *(No word limit- please expand the text box as necessary).*

<u>What type of data about the red deer in Borrowdale do you consider important for selecting impact management approaches?</u>

Nine respondents requested for a combination of both industry standard and academic, statistical, methods. Two participants suggested industry standard methods only, and no votes cast for statistical methods only.

Selected participant comments:

- "Statistical methods have a place alongside industry methods as another management tool" and "it would be a shame to stop using" industry methods. Further, "a blend of the current industry standard techniques in addition to a certain number of statistical methods will give the most valuable information in terms of all the data required to best manage the deer population".

- "Industry standard methods should be used as a default given that they are quicker to apply, more cost effective and can be used by the people 'on the ground'" but "they may benefit from being tested and validated by Statistical Methods, in some cases periodically". Also, statistical methods may help to address "situations where industry standard methods are not providing the desired level of detail, or where there is disagreement between parties".

- "Standard methods help provide consistency and replicable data which can be analysed at different geographical levels (nationally, county, valley/landscape) to see the 'big picture'". There was concern from one participant as to whether statistical methods are focussed around "money"

- "Given the restrictions due to finance, industry methodology will suffice for a number of years" as "a ten-year data bank should suffice in answering most of the major questions".

Q4. Do you have any comments regarding the objectives guiding management and how deer feature within these objectives? *(No word limit- please expand the text box as necessary).*

Section 3.

Identifying the acceptability of methods of managing the red deer population and applying this across a landscape-scale management plan for Borrowdale and Thirlmere.

Q5. All 13 participants from the first round of the questionnaire surveys agreed that deer required management in the Borrowdale and Thirlmere region, with five methods identified by participants. Please add your comments about each method with regards to managing the Borrowdale and Thirlmere red deer population in the table below.

Method Theme	**Please add your comments about each method on their ability to manage this red deer population** *For example, you could consider the plausibility, effectiveness, and acceptability of the method, or any other comment about the method you may have.* *(Please expand the text boxes as necessary).*
Lethal Control *I.e., culling*	
Translocations *I.e., transporting red deer to elsewhere*	
Exclusion *I.e., preventing deer accessing places or plots*	
Predator Reintroductions *I.e., reintroducing wolves and/or lynx into the Lake District*	
Breeding Control *I.e., contracepting / sterilising wild red deer*	

Q6. Please comment as to whether the method would likely assist (and why) in meeting the established landscape objectives; summarised as 'restoring and regenerating resilient habitats and protected sites to assist in natural flood management and accommodate sustainable and healthy populations of wildlife'.

For the methods you said yes or unsure, please state how success or failure could be measured in terms of meeting the established objectives.

Method Theme	**Would this method assist in meeting the stated landscape objectives?** *(Please explain your answer)*	**How could you measure the success and/or failure of this method to meet the stated objectives?** *(Answer for all methods that you selected yes or unsure)*
Lethal Control *I.e., culling*	*Yes / No / Unsure (delete as necessary)*	
Translocations *I.e., transporting red deer to elsewhere*	*Yes / No / Unsure (delete as necessary)*	
Exclusion *I.e., preventing deer accessing places or plots*	*Yes / No / Unsure (delete as necessary)*	
Predator Reintroductions *I.e., reintroducing wolves and/or lynx back to the Lake District*	*Yes / No / Unsure (delete as necessary)*	
Breeding Control *I.e., contracepting / sterilising wild red deer*	*Yes / No / Unsure (delete as necessary)*	

Q7. What should be done by those formulating the impact management strategy if the evidence shows that the selected methods and targets are not meeting the stated landscape objectives? *(No word limit- please expand the text box as necessary)*

END OF QUESTIONNAIRE – THANK YOU FOR YOUR PARTICIPATION

Appendix C. Chapter 3 Supplementary Tables

Table C1. Measurements of the red deer population discussed, and decisions made, by workshop participants.

Measurement of Deer the Population	Accepted or Rejected by participants for the adaptive impact management framework	Key Workshop Participant Discussion Points
Animal Welfare Health of individual animals across the herds.	Accepted (in part)	Good red deer welfare would be considered the result of a "healthy" **Participant 7** and "sustainable" **Participant 2** herd. Welfare status is not necessarily a "key information need" as it should "come out of adequate management" **Participant 5**. Welfare is "inextricably linked to population size" **Participant 3**, and therefore an "overpopulation of deer [would have] poor welfare" **Participant 1**.
Deer Impacts Impacts from deer to woodlands and to open hill habitats. Using quick industry-standard assessment methods.	Accepted	Monitoring of deer impacts should be the "joint number 1 [data requirement] with population size" **Participant 6**. But, if resources are limited, "focus on impacts instead of population size" **Participant 5** as "impacts vary and are more important" **Participant 7**. Further, "cull targets [can be set using] impact assessments" **Participant 4** and "helps fulfil objectives and information on impact" **Participant 2**.
Population Size Needs to be reliable, whether a census or sampling method. A combination of on-foot census counts and drone census surveys	Accepted	A reliable estimate of red deer population size should be the "joint number 1 [data requirement] with deer impacts" **Participant 6**. Knowing a population size is an "important conversation key" **Participant 5** as "stakeholders want to know you have a handle" on the population **Participant 2**. The "granularity of the data [is important]" for confidence, "[such as] drone photographic evidence" **Participant 1**.
Population Dynamics	Accepted (in part)	Understanding population dynamics is key to informing management actions, with a "need to understand calving rate" **Participant 6** to focus strategy. This enables

Understanding the population sex ratio, fecundity, mortality etc...		targeted culling efforts as, for example, it is "pointless [only] culling stags if the fecundity is really high" **Participant 5** as the female population drives population growth. These data can be extrapolated from population size data.
<u>Range and Movement</u> Defined as how the herd uses the landscape. Can be extracted from camera trapping and drone surveys.	Rejected	Range and movement of deer was not considered to be important in decision-making "because of good collaborations in the area" **Participant 1** enabling effective communication of deer sightings. However, such information could be important for "a piecemeal [fragmented] landscape" **Participant 3** and "could be used to pressure landowners" that do not work collaboratively throughout landscapes **Participant 2**. Information could be extracted from population estimate data, for example "drone surveys give a good idea of where the deer are" **Participant 7**.
<u>Collaborations</u> Who is involved in management decision-making across the landscape.	Rejected	It was agreed by all participants that collaborations are "important" to good deer management, but do not count as "information" required for management in this landscape.
<u>Traffic Collisions</u> Road traffic collisions involving deer.	Rejected	The requirement for information on traffic collisions to be included in the Adaptive Management Framework was rejected by all participants without further discussion.

Table C2. Methods of red deer population control discussed, and decisions made, by workshop participants.

Method of Deer Population Control	Accepted or Rejected by participants for the adaptive impact management framework	Key Workshop Discussion Points
Lethal Control Enacted through culling deer by rifle using professional deer managers.	Accepted	Lethal control was agreed to be the key method of managing deer by the participants as it is the only method that “reduces deer [population size] and moves them” away from sensitive areas **Participant 6**. Further, lethal control “complements” any exclusion efforts (below) for deer in sensitive areas **Participant 5**.
Exclusion and repellence Adjusting how deer are able to move through the landscape. This can be enacted through fencing, tree tubes, and chemical repellents.	Accepted	Large scale deer fencing is “expensive for landowners” to use, with challenges in planning permission from “common ground” used by the general public, and there are efforts to “reduce” fencing **Participant 2**. Further challenges to planning permission come from “opposition [by people] to visual barriers” on the open hillside and in woodlands **Participant 1,** and the expense for “maintenance” of fencing and tree tubes **Participant 7**. Further, large scale “deer fencing [does not avoid the] need to shoot deer [that end up] within the fences” **Participant 3**. Any “exclusion [effort] should be strategic” and also assist with controlling sheep **Participant 6** as when the “sheep are gone” there are “signs of [natural] recovery” **Participant 2.** Stakeholders would also like to “trial a deer repellent” as an additional “tool” for management **Participant 1.**
Predator reintroductions Enacted by releasing formerly extirpated predators, such as European grey wolf (*Canis lupus*), Eurasian lynx (*Lynx lynx*), and	Rejected	Reintroducing predators was rejected as doing so is “not practical in England” as the “ecosystems are unable to support them” **Participant** 5. Participants agreed there can be no attempt to reintroduce predators without the “healthy, balanced, landscape needed” **Participant 3** to support such a change in the ecological communities. It was agreed that reintroducing such predators “would be nice [but there is] not a big enough area [with] connected habitats” to be suitable **Participant 4**.

Eurasian brown bear (*Ursus arctos*)		
<u>Fertility control</u> Medically adjust female fertility using immunocontraceptives.	Rejected	The use of fertility control was rejected by participants as the method “does not fit with the ethos and objectives of natural processes” in the area **Participant 2**. Further, the method was not considered effective in the short term as it “does not [immediately] reduce a population size” **Participant 1**, thus would not address immediate impact concerns.
<u>Translocations</u> The capture and relocation of deer from their home range to another area.	Rejected	Translocations were rejected as a method for multiple reasons. The key reason was the high levels of “difficulty obtaining licences for wild deer in the UK” from Natural England **Participant 3**. There were further concerns raised of “significant welfare implications” to deer from translocations that have a “low success rate” related to “high [levels of] mortality” **Participant 1**. **Participant 4** stated that “Shooting is [considered to be] more humane” than stress related deaths, which was agreed by all participants.

Appendix D. Chapter 3 Workshop Agenda

9:30 Arrival

9:45 Welcome

10:00 Session 1: Vision, Objectives, and Information Needs

11:00 Coffee break

11:15 Session 2: Methods for managing red deer

12:15 Lunch break

13:15 Session 3: Adaptive deer management

14:15 Coffee break

14:30 Feedback and summary session

15:00 End

Appendix E. Chapter 5 Supplementary Materials

Table E1. Spearman's rank correlation outputs evaluating intensive and index impact indicators across woodland and open hill habitats.

Method	Indicator	df	r_s	p
Woodland	Broken Stems	16	0.52	0.152
	Fraying/Thrashing	16	0.416	0.266
	Browse Lines	18	0.524	0.12
	Coppice/Basal/Saplings	18	0.703	<0.05
	Bramble	18	0.787	<0.05
	Grazing	18	0.742	<0.05
Open Hill	Ling form	32	0.728	<0.05
	Ling broken stems	32	0.267	0.284
	Ling browsed	32	0.69	<0.05
	Erica tetralix	12	0.632	0.126
	Erica cineria	18	0.486	0.154
	Blaeberry	32	0.461	0.053
	Moss cover	18	-0.355	0.314
	Bare ground	18	0.382	0.277
	Sward Height	22	0.866	<0.001
	Palatable grazing	22	0.65	<0.05
	Unpalatable grazing	22	0.604	<0.05
	Nardus stricta	20	0.717	<0.05

Table E2. GLMM output for changes to index indicators over three years for DAI and Putman Method.

Method		Estimate	Std. Error	Z	P
DAI Activity	(Intercept)	0.843	0.376	2.243	<0.05
	Year	-0.124	0.178	-0.699	0.485
DAI Impact	(Intercept)	0.642	0.351	1.832	0.067
	Year	0.058	0.167	0.346	0.729
Putman Impact	(Intercept)	1.114	0.264	4.217	<0.001
	Year	-0.062	0.125	-0.497	0.619