TREE PROTECTION PROJECT Work Package 1

Impacts of mammals on trees and tree protection methods pertinent to English treescapes - a systematic literature review for Forestry Commission

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SUMMARY

We conducted a systematic literature review as part of a Forestry Commission (FC) contract to provide good practice technical guidance to manage impacts of mammals on trees, woods, establishing woodlands and treescapes. The review examined both the positive and negative impacts of mammals and existing tree protection options and best practice. Using the PRISMA ('Preferred Reporting Items for Systematic reviews and Meta-Analyses') framework (Page et al., 2021), we systematically searched for relevant scientific, peer-reviewed literature. We also conducted targeted organisational searches for grey literature relevant to the objectives within an English context. We then implemented a thematic tagging process to facilitate the synthesis of findings across both types of literature, whilst building a categorised resource bank. Following exclusions, we identified and tagged 281 scientific literature sources and 218 grey literature sources. We identified eight key species or functional groups of mammals (beavers; small mammals; grey squirrels; lagomorphs; pigs and wild boar; deer; livestock including feral sheep and goats and bison; and horses and ponies) that have distinct tree protection approaches available, relating to their ecology, behaviour, type of damage caused and legal restrictions. We also developed a novel ecological framework to categorise tree protection methods according to their strategy of intervention within mammal-tree interactions and applied it to each species/functional group. Finally, we provide an extensive synthesis of the literature for each species or group, detailing literature relating to ecosystem services, damage to trees, identification of that damage, and resultant protection methods identified. This document thus provides a foundation for developing the practitioner-focussed technical guidance.

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CONTENTS

SUMMARY 1

ACKNOWLEDGEMENTS 2

1 RATIONALE AND OBJECTIVES 4

1.1 Rationale for review 4

1.2 Objectives and scope of review 4

1.3 Document structure 4

2 METHODS 4

2.1 Framework and search approach 4

2.2 Search terms for primary scientific literature 7

2.2.1 Search category: Mammal taxa 7

2.2.2 Search category: Damage 8

2.2.3 Search category: Tree specifier 8

2.2.4 Search category: Geography 8

2.2.5 Search category: Protection 9

2.2.6 Search category: Confusion terms 9

2.3 Search combinations 10

2.4 Search databases and dates 11

2.5 Targeted search for grey literature 12

2.6 Search result management, exclusion, and tagging 12

2.6.1 Workflow and exclusion criteria 12
2.6.2 Tagging of relevant sources 14
2.6.3 Inter-observer consistency of tagging 14
3 RESULTS 16
3.1 Inclusion and exclusion 16
3.2 Grey literature sources 17
3.3 Summary of articles by year, type, geography, taxa and damage 18
3.4 An ecological framework for tree protection approaches 21
3.4.1 Applying the framework to mammal functional groups 24
4 DISCUSSION 33
4.1 Unintended consequences of tree protection methods 33
4.2 Considering carbon costs of tree protection methods 34
4.3 Format of literature 34
4.4 Synthesis of damage and protection approaches for functional groups 35
5 APPENDICES 36
5.1 Notes on the appendices 36
5.1.1 Beaver (Family Castoridae) 36
5.1.2 Grey squirrel (Family Sciuridae) 41
5.1.3 Small mammals (Families Cricetidae, Muridae and Gliridae) 51
5.1.4 Lagomorphs: Rabbits and hares (Family Leporidae) 59
5.1.5 Pigs and wild boar (Family Suidae) 65
5.1.6 Deer (Family Cervidae) 68
5.1.7 Livestock: Sheep, goats, cattle, and bison 83
5.1.8 Horses and ponies (Family Equidae) 97
6 REFERENCES 100

1 RATIONALE AND OBJECTIVES

1.1 Rationale for review

A call to action in The UK Government's England Trees Action Plan 2021-2024 included "We hope stakeholders will… access more and better advice and guidance on establishing and managing trees and woodlands" (The UK Government, 2021). The Forestry Commission (FC) is seeking good practice technical guidance to manage impacts of mammals on trees, woodlands and treescapes. The aim is to outline both the positive and negative impacts of mammals and develop a range of effective mammal mitigation and management options for woodland owners and managers. A review of literature (Work Package 1) is required, which will feed directly into a new technical guidance document (Work Package 2) to be made publicly available.

1.2 Objectives and scope of review

The review has two primary objectives set by FC:

Objective 1 – identify both the positive and negative impacts of mammals on trees, woods, establishing woodlands and treescapes; and

Objective 2 – identify tree protection options and best practice.

In terms of scope, it is requested that relevant grey literature (such as industry standards) and non-UK evidence is also considered. For the latter, the scope of non-UK evidence of impacts will be restricted to species found in the UK or with potential to be re-introduced to the UK or used in the UK for conservation purposes. Tree protection methods may consider approaches used for species not found in the UK but with functionally similar impacts.

1.3 Document structure

The document consists of a main text (Sections 1-4: Introduction, Methods, Results and Discussion) that summarise what we did, quantifies what we found in terms of literature and summarises visually tree protection methods identified for mammal groups with a simple measure of evidence for their effectiveness.

A longer synthesis consisting of detailed text citing sources in the literature under themes is in the appendices (Section 5). All references have been provided at the end of the document (Section 6) and are either open access or can be made available on request. A larger, tagged resource bank in (BibTex and MS Excel formats) have been submitted alongside the document.

2 METHODS

2.1 Framework and search approach

It was critical to consider primary scientific sources (typically peer-reviewed journal articles and book chapters, and theses) as well as grey literature, consisting of various types of technical guidance, news articles, blogs, web resources or unpublished research reports. These require different approaches, outlined in Table 1. Scientific literature is typically indexed and consistently searchable via title, abstract, and keywords, while grey literature has more variability in format, metadata, and labelling.

Considering the objectives outlined by FC we recognised that specific elements from Objective 1 "negative… impacts of mammals on trees, woods, establishing woodlands and treescapes", namely specific damage to trees, and Objective 2 "existing tree protection options and best practice" should be considered together, since the latter is a response to the former and some literature was likely to refer to both.

Table 1. Search approaches taken for each type of literature. Note that it was possible occasionally to find grey literature through systematic review and primary scientific literature via targeted organisation/website searches.

Type of literature	Search approach
Primary scientific literature	A PRISMA systematic review approach was taken to ensure transparency and repeatability of the method (Page et al., 2021).
Grey literature	We identified organisations and websites that were highly relevant to Objectives 1 and 2 and searched their sites for relevant literature, similar to the approach of Bernes et al. (2015).

A large element of the project specification was to focus on "damage", and this is referred to in the context of "preventing tree establishment, healthy growth or good timber quality", "preventing the development of a structurally diverse shrub layer", "significantly modifying the composition of the ground layer". It also stated specifics such as "browsing", "bark stripping", "gnawing" and "rubbing". In addition, "protection" and "mitigation" are emphasised. The concept of damage is relatively easy to define. In their three-part review of mammal damage to trees Gill (1992a, 1992b, 1992c) defines damage as: *"Injury to trees in the form of tissue removal (leaves, bark, flowers, shoots, buds etc.). It does not necessarily imply economic loss."*. Protection and mitigation in this context are also relatively easy to define, as a deliberate action intended to prevent or reduce damage.

On the other hand, "positive impacts of mammals on trees, woods, establishing woodlands and treescapes" operate along a spectrum of plant-animal interactions. These might be positive or negative depending on (a) a value judgement (for example tree mortality might be negative from a commercial point of view or positive from a conservation point of view, if it creates deadwood), (b) context (for example browsing deer might have maximum biodiversity benefits at intermediate population levels). In a review of impacts of ungulates on vegetation, Reimoser and Putnam (2011) argue that "*damage caused by large ungulates, whether to commercial or conservational interests merely represents the extreme expression of a whole suite of general changes, subtle deflections in the structure and dynamics of natural communities in response to herbivory*" and that "*grazing and browsing, as well as trampling, or rooting… , may all have many positive, facilitative effects within natural or managed communities, as well as a potential for causing damage*". In addition, positive impacts of taxa/species on broader environments are typically framed in terms of ecosystem services or functional roles (Lacher et al., 2019) and the Project Management Plan specifically indicates the objective to understand mammal impacts on the "wider ecosystem".

We looked for clear evidence of (and mitigation against) "damage" but also gathered information from our search results on broader impacts on ecosystem services or functions. This framework is outlined in Figure 1, and the search and tagging approach we have taken in line with this framework is indicated in Table 2.

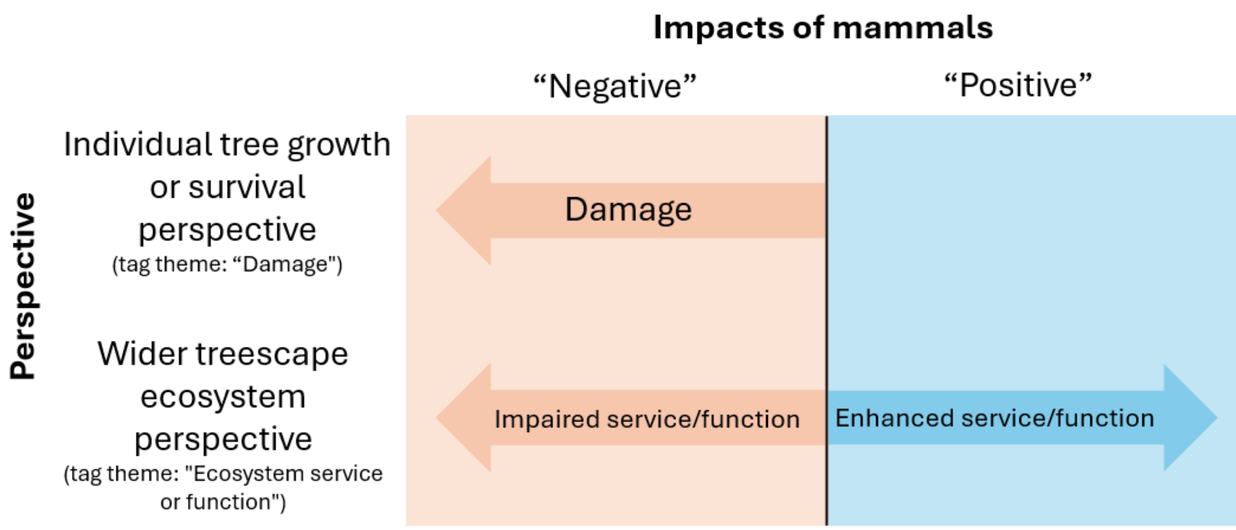

Figure 1. Framework in terms of considering "positive and negative impacts of mammals on trees, woods, establishing woodlands and treescapes" (as per Objective 1). From the perspective of individual tree growth or survival or health, we consider impacts will be predominantly "negative" and can be referred to as "damage" (sensu Gill, 1992a).From the wider treescape ecosystem perspective impacts can be framed in terms of ecosystem services or functions and may exist along a continuum that may depend on value judgement or management context. Note the clear line between negative and positive impacts is not always clear as shown, and may depend on management objectives.

Table 2. How different aspects of the Project Objectives were dealt within the framework of searching and tagging of literature.

Aspect of objectives	Dealt with via
Objective 1 - "*negative… impacts of mammals on trees, woods, establishing woodlands and treescapes*" Objective 2 - "*existing tree protection options and best practice*"	Literature searching targeted to tree(scape) damage and protection and tagging literature according to 'Damage' and 'Protection' tags*
Objective 1 - "*positive impacts of mammals on trees, woods, establishing woodlands and treescapes*"	Initially tagging literature with 'Ecosystem service or function' tags (inclusive of positive, negative or nonlinear effects). When preparing the technical

	guidance document, we will conduct targeted search to fill any gaps as necessary.

**Tags and tagging are explained in more detail below.*

2.2 Search terms for primary scientific literature

Search terms were split into a number of overarching categories. Boolean operators are the terms 'AND', 'OR' and 'NOT' that are recognised by search functions of most databases or search engines that help to narrow or broaden searches. Categories are connected by the operator 'AND' (or in some cases 'NOT', described later) which aimed to narrow the search by ensuring search results will tend to cover all of categories considered necessary. For example, we might want search results to pertain to a restricted set of mammal taxa AND refer to some kind of impact AND that impact be related to trees, forests or woodlands AND the study to be geographically relevant (e.g. UK and areas ecologically similar to the UK). However, within those categories were specific alternative search terms that are connected by the operator 'OR', which broadens the search within a theme, recognising there may be different terminology used for the same concept or that studies address different aspects of a theme. For example, within the mammal 'Taxa' theme, we may want to consider studies that refer to 'mammals' OR 'herbivores' OR 'deer' OR 'lagomorph' etc. Allowing the search to locate sources referring to one of a range of specific taxa, or to several, or more generally to herbivores or mammals.

For primary scientific literature, the search engines used lemmatisation (Web of Science - Clarivate, 2024) and stemming (Google Scholar – Catriona Robertson, Subject Librarian pers. comm.) whereby they automatically find variant spellings and plurals of words. In addition, Web of Science automatically finds US and UK spelling differences (e.g. 'grey' vs 'gray'). As such we only included singular forms of words, but to err on the side of caution, for terms that could have several alternative spellings such as 'fence' and 'fencing', or for irregular plurals such we used the asterisk truncation symbol, i.e. 'fenc*'. We also included singular and plural where the plural was irregular, such as 'mouse' and 'mice'. For terms that could form part of several useful terms such as 'forest' ('forestry', 'afforestation', 'agroforestry') we used wildcard notation '*forest*'. For exact phrases sought, for example United Kingdom (and not wanting to search separately for United and Kingdom) we used quotation marks, i.e. "United Kingdom".

Term inclusion/exclusion within categories was a balance of relevance to the scope and objectives of the review and some pilot-stage trial and error in terms of the balance of apparently relevant and irrelevant results produced (exclusion criteria for non-relevant documents are explained below). Here we present the different categories and reasons for inclusion/omission of terms within them.

2.2.1 Search category: Mammal taxa

UK mammals were searched from the Mammal Society's full UK species list (Mammal Society, 2024) and we extracted all species with known impacts on trees and checked and modified with input from the Project Steering Group (Box 1). We also cross checked against a Forest Research resource (Forest Research, 2024a) to look at possible species that are known to cause tree damage in the UK. European mole *Talpa europaea* was excluded in Steering Group discussion, and red-necked wallaby *Macropus rufogriseus* was excluded as it is now considered extinct in England (People's Trust for Endangered Species, 2024). General terms 'mammal' and 'herbivor*' were used to pick up any multispecies studies/reviews that may not mention species in the title, abstract or key words and we included domestic grazing species as well as species used for conservation grazing.

Box 1. Search terms included within the 'Mammal taxa' category

mammal OR vertebrate OR herbivor* OR lagomorph OR rabbit OR hare OR rodent OR squirrel OR beaver OR vole OR mouse OR mice OR badger OR boar OR pig OR deer OR muntjac OR cervid OR goat OR sheep OR cow OR cattle OR buffalo OR bison OR horse OR pony OR ponies

2.2.2 Search category: Damage

General terminology such as 'negative', 'impact' etc. were found to generate too many results, as such terms are likely to be included in a very wide range of scientific papers and refer to a wider range of contexts. Based on trials we found that use of very specific terms that referred to negative processes that mammals have on trees, for example those indicated by Forest Research (2024a) produced relevant papers. The terms "dig*" and "rooting" which might refer to impacts of e.g. wild boar *Sus scrofa* pulled across a very large number of apparently irrelevant documents.

Box 2. Search terms included within the 'Damage' category

damage* OR brows* OR graz* OR trampl* OR bioturb* OR poach* OR "bark strip*" OR "strip bark" OR retard

2.2.3 Search category: Tree specifier

So that papers extracted consider the above impacts on trees and treescapes *per se*, we specified terms that referred to woods, forests and treescapes (Box 3). We also included terms "shrub", "scrub" and "rewild" to enable inclusion of documents relevant to include developing woodlands.

Box 3. Search terms included within the 'Tree specifier' category

tree OR *forest* OR wood* OR carr OR shrub OR scrub OR "rewild" OR coppic*

2.2.4 Search category: Geography

Biomes and climatic regions relevant to England were considered (e.g. ‘temperate’, ‘oceanic’) but discarded following pilot searches. The temperate biome includes areas from several continents including the southern hemisphere, and early search attempts were geographically too broad. Geographically, as well as specifying ‘England’, ‘UK’ and ‘”United Kingdom”’, we specified the geographical regions close and/or including England. Definitions of European regions vary, with the UK being considered part of Northern Europe (United Nations, 2024) and Western Europe (European Union, 2024) respectively, so both were included (Box 4). We also included “Central Europe” since some countries considered part of Western Europe such as Germany are sometimes considered to be Central Europe. In addition, some species that have become extinct from Great Britain such as European bison *Bison bonasus* and Eurasian beaver *Castor fiber* and are being reintroduced or used for conservation still have extant populations in Central Europe so studies from this region are likely to be pertinent.

Box 4. Search terms included within the ‘Geography’ category

UK OR “U.K" OR “United Kingdom” OR Britain OR British OR Engl* OR “Northern Europe” OR “Western Europe” OR “Central Europe”

2.2.5 Search category: Protection

Following some initial exploration, a wide variety of terms relating to tree protection were included (Box 5) to account for the array of approaches that might be used, inclusive of specific and general terminology.

Box 5. Search terms included within the ‘Protection’ category

"tree protection" OR prevent* OR fenc* OR exclusion OR exclud* OR “tree tube” OR “tree shelter” OR “tree guard” OR spray OR treatment OR repel* OR deter OR deterrent OR control* OR diversion* OR *palatab* OR shoot* OR trap* OR poison* OR “damage assessment” OR “risk assessment” OR “impact assessment” OR predator OR predation OR “biological control” OR “browsing assessment” OR "nurse plant" OR "nurse shrub" OR contracept*

2.2.6 Search category: Confusion terms

During early searches, we noticed that several terms generated a relatively large number of non-relevant results. For example, deer mice *Peromyscus spp.* can damage trees in North America and have thus been the focus of research. Such outputs that will tend to appear given our search terms “deer” and “mice” (Box 6). Any such terms, shown in Box 6 were excluded from searches using the operator ‘NOT’ for Web of Science, although not for Google Scholar which did not appear to recognise this operator.

Box 6. Search terms included within the 'Confusion terms' category. Searches A and B are explained below.

Search A: "British Columbia" OR "New England" OR "deer mice" Search B: "British Columbia" OR "New England" OR "seed trap" OR insect OR arthropod OR invertebrate OR beetle OR aphid OR gastropod OR caterpillar OR leafminer OR weevil

2.3 Search combinations

We anticipated that there might be studies that refer to mammal damage yet not to protection or mitigation against that damage. Similarly, there might be studies that refer to tree protection approaches but did not refer specifically to any species in the title, abstract or keywords (which we targeted in Web of Science searches, see below). As such we carried out two different searches (A and B) and pooled the results. This allowed us to include both types of study, plus studies that referred both to species causing damage and protection method (Figure 2). This was achieved by searching with different sets of search categories and resultant search strings (Table 3) and then pooling the results.

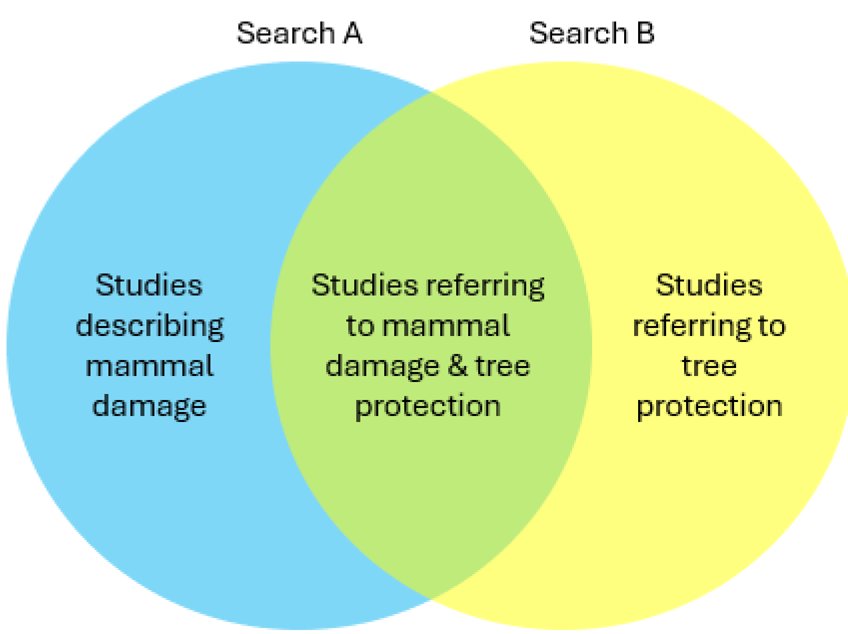

Figure 2. Demonstration of search approach whereby we combined out two searches (A and B) to target a broader range of studies.

Table 3. Details of final search strings used: (a) mapping search against categories, and (b) actual search terms. Note that for Web of Science each category is put in a separate search box separated

by AND or NOT operators, while for Google Scholar the whole string is placed in the search. For Google Scholer searches, the NOT operator appeared to not be recognised so Confusion terms were not included.

(a)

Search category	Search A	Search B
Mammal taxa		
AND Damage		
AND Tree specifier		
AND Geography		
AND Protection		
NOT Confusion terms	(Web of Science only)	(Web of Science only)

(b)

Search	Full search string
A	(mammal OR vertebrate OR herbivor* OR lagomorph OR rabbit OR hare OR rodent OR squirrel OR beaver OR vole OR mouse OR mice OR badger OR boar OR pig OR deer OR muntjac OR cervid OR goat OR sheep OR cow OR cattle OR buffalo OR bison OR horse OR pony OR ponies) AND (damage* OR brows* OR graz* OR trampl* OR bioturb* OR poach* OR "bark strip*" OR "strip bark" OR retard) AND () AND (UK OR "U.K" OR "United Kingdom" OR Britain OR British OR Engl* OR "Northern Europe" OR "Western Europe" OR "Central Europe") *NOT ("British Columbia" OR "New England" OR "deer mice")*
B	(damage* OR brows* OR graz* OR trampl* OR bioturb* OR poach* OR "bark strip*" OR "strip bark" OR retard) AND (UK OR "U.K" OR "United Kingdom" OR Britain OR British OR Engl* OR "Northern Europe" OR "Western Europe" OR "Central Europe") AND ("tree protection" OR prevent* OR fenc* OR exclusion OR exclud* OR "tree tube" OR "tree shelter" OR "tree guard" OR spray OR treatment OR repel* OR deter OR deterrent OR control* OR diversion* OR *palatab* OR shoot* OR trap* OR poison* OR "damage assessment" OR "risk assessment" OR "impact assessment" OR predator OR predation OR "biological control" OR "browsing assessment" OR "nurse plant" OR "nurse shrub" OR contracept*) *NOT ("British Columbia" OR "New England" OR "seed trap" OR insect OR arthropod OR invertebrate OR beetle OR aphid OR gastropod OR caterpillar OR leafminer OR weevil)*

2.4 Search databases and dates

We carried out scientific literature searches in two complementary databases that contain a very large number of scientific sources: Web of Science and Google Scholar. Web of Science is a subscription service which Edinburgh Napier University has access to, which indexes the world's leading peer-reviewed journals across sciences and social sciences (Harvard Library, 2024). Google Scholar is a broader search engine that also includes theses, technical reports and documents that academics/researchers themselves include as scientific outputs (pers. obs.). Searches were carried out in slightly different ways depending on database (Table 4).

Table 4. Databases, search dates and extraction notes for those databases. Note the BibTex format is a widely used bibliographic format that allows cross communication between search engines and reference management software.

Database	Date of search(es)	Notes

Web of Science	Search A 18 Jun 2024 Search B 17 Jun 2024	The search is only done on "Topic" (searches title, abstract, keyword plus, and author keywords). Each category of terms is included in a separate search box, with AND or NOT operators between. All results are extracted to reference manager via BibTex format.
Google Scholar	Searches A and B 2 Jul 2024	A full search string is used and given very large returns that Google Scholar gives (c. 900,000). We looked at first 200 results (as per Bernes et al., 2015) plus then scanned subsequent pages of 10 items beyond 200 and stopped when we reached the first page of 10 items where no relevant result was found. Results were first scanned by title and abstract, and any relevant or potentially relevant references were extracted into BibTex format.
Google	various Jun-Oct 2024	Used *ad hoc* to identify documents, organisations, and websites of relevance, particularly for grey literature.

2.5 Targeted search for grey literature

Grey literature is not formally indexed and can take a wide variety of formats. As such, we took a targeted approach by scouring websites of specialist environmental, forest and land organisations for relevant grey literature (see Results). We also took a 'snowball sampling' approach whereby if searching one organisation led us to another we had not yet identified, we further searched that organisation's website if relevant. We also included sites that compile research theses and dissertations which are not published but may contain relevant research findings. These were ProQuest database of theses and dissertations, EThOS, British Library Electronic Theses Online Service, and Open Access Theses and Dissertations.

2.6 Search result management, exclusion, and tagging

2.6.1 Workflow and exclusion criteria

Search results were managed in the reference manager Zotero (Corporation for Digital Scholarship, 2024). Broadly speaking, the process of literature review involves (i) search (see above), (ii) exclusion/filtering of non-relevant results, (iii) tagging of relevant results, and (iv) synthesis of findings utilising tags to search for literature with specific themes and/or quantify patterns in the findings. The workflow of exclusion via filtering and tagging is outlined in Figure 3. Often there are several reasons a paper would be excluded and so we operated a hierarchical filtering process whereby we noted only the first reason a resource could not be included in the synthesis. For the purpose of transparency in our review, records of excluded papers were kept. This allowed us to quantify the number of exclusions and reasons for this, and also quantification of effectiveness of search strings used. For included resources, PDFs (or website links for web resources) were retained in a resource bank for later submission to FC. Most PDFs were available via at least one of several routes including the paper being open access, behind a paywall but available via Edinburgh Napier University subscriptions, available directly from university or personal repositories via

Google Scholar or ResearchGate, or requested via ENU library using the inter-library loan process. A small number of resources (typically a relatively old scientific paper or a book with limited print run) was unavailable through any of these routes.

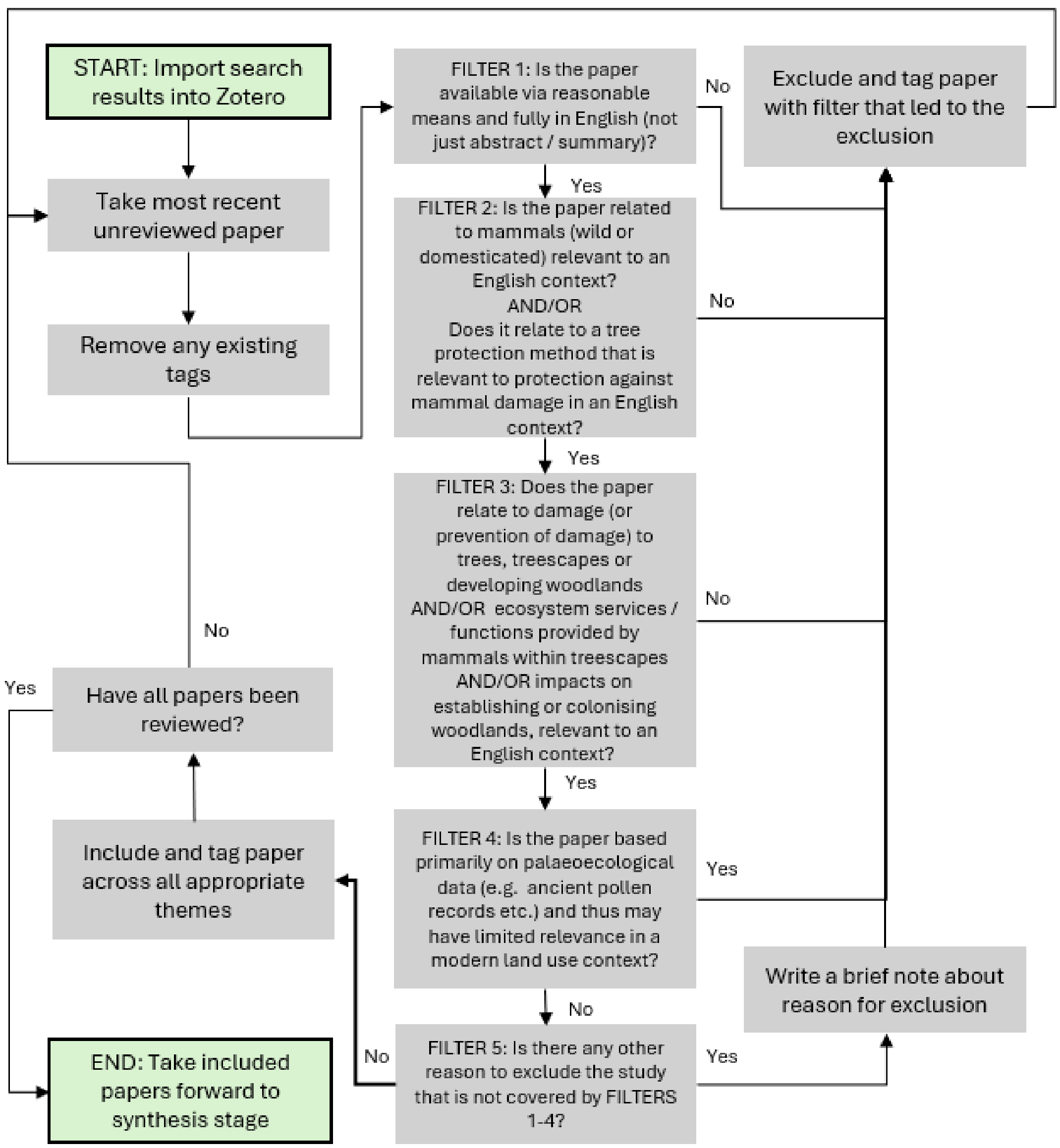

Figure 3. Workflow of exclusion and tagging of resources outputted from searches (see above). The exclusion process (middle column) was via a series of filters. If a resource passed each filter, then it was included and tagged across a series of themes (discussed below). The final output is a set of relevant literature (scientific and grey) that can be taken forward to the synthesis stage.

Papers discussing mammal effects on woody plant encroachment onto open habitats were included if they could be regarded as “developing woodlands”. For example, colonisation by shrubs/trees in this habitat type might be allowed to persist and undergo succession, either transforming the whole habitat or as a mosaic of tree/shrub-dominated vegetation with open habitats. If the paper focused on open habitats which are highly valued and on preventing encroachment to maintain those open habitats in perpetuity, then these were not regarded as "developing woodland". Such papers were excluded unless they contained other information relevant to planted/developing woodland e.g. the relative palatability or susceptibility to grazing/browsing of different tree/shrub species.

Despite geographical specifiers, occasional geographically distant studies will appear in results. As a general rule if the tree species affected are used widely in an English context (such as Sitka spruce *Picea sitchensis*), the mammal taxa were in our list or were at least functionally very similar (such as certain species of deer), and the study was in a biome that overlapped somewhat climatically with the UK (such as northern temperate or boreal climates) we would include them and acknowledge their geographical limitations in the synthesis.

2.6.2 Tagging of relevant sources

The main functions of tagging are to allow (i) some quantification of themes that are present within the literature, and (ii) subsequent targeted searching of identified relevant literature for the purpose of synthesis in this document or during the technical guidance document preparation. Tags are designed to be non-mutually exclusive using a "check/tick all that apply" (CATA) approach. For example, a resource can be tagged as "Damage - Browse foliage/buds" and "Damage - Bark damage" if both are reported in the paper. For tags under the 'Damage', 'Protection' and 'Ecosystem service or function' themes, sources were tagged to that theme if that theme is reported on, or discussed in the paper, regardless of magnitude or direction of any effect found. We did this because we wanted to account for lack of effects or different directions of effects during the synthesis, nuanced or context-dependent effects, or apparent received wisdom in the case of grey literature. Tagging was done across a range of themes which are described in Table 5. The full list of tags within themes is large and fluid since tags may be combined during synthesis (or new tags added) as we learn more about how impacts and protection methods are related or differentiated. Frequently, papers exported in BibTeX format contain tags added by the publishers (such as keywords etc.); to avoid potential confusion these were deleted prior to our tagging.

2.6.3 Inter-observer consistency of tagging

Time excluded the possibility for double assessment of every relevant document. Given two researchers were involved in tagging (PJCW and JM), however, it was important to ensure we were tagging sources consistently. We thus carried out an initial cross-checking exercise to calibrate tagging between the two observers. To do so, we both looked independently at 50 sources and discussed any occasions where we differed in how we would apply tagging to papers. This was done early in the tagging process when we had tagged sufficient sources for a comparison.

Table 5. Tagging themes and function. Note some other tag themes were used for internal workflow management purposes but are excluded here.

Theme	Function
Status	Used for document management including current status of paper e.g. included in review, excluded (with reason) and yet to tag.
Document type	Indicates if the document is e.g. a review paper, based on field research, a technical guide etc.
Mammal	Indicates what relevant mammal taxa are discussed in a paper, either specific species or broader taxa (e.g. deer in general, voles in general).
Woodland/treescape types	Used to broadly indicate the types of woodlands or treescapes discussed, either quite broadly e.g. “broadleaved”, “seminatural” etc., or specific e.g. “coppice with standards”, “wood pasture”. Also included tags relating to age and stage, e.g. “mature”, “colonising” or objective, e.g. “rewilding”, “commercial”.
Damage	Referred to specific elements of damage to trees sensu Gill (1992a) (see Figure 1). Also noted when economic damage was quantified (absolutely or relatively).
Protection	A range of direct, e.g. “tree tubes” or indirect e.g. “cover manipulation” methods that have been proposed or tested as tree protection methods.
Ecosystem service or function	Includes wider impacts on treescape ecosystems by mammals including those outlined by Lacher (2019), whether enhanced/impaired (see Figure 1).
Potential as a case study	Studies that report systems or methods that might serve as illustrative case studies for the Technical Guidance Document (Work Package 2).
Country	Also notes where a study is not geographically specific or carried out on a continental scale (e.g. Europe).
Organisation	Relevant to grey literature, which organisation/authority is the resource provided by (from those in Table 6).

3 RESULTS

3.1 Inclusion and exclusion

A PRISMA flow chart of searches, exclusions and inclusions in the final literature is shown in Figure 4.

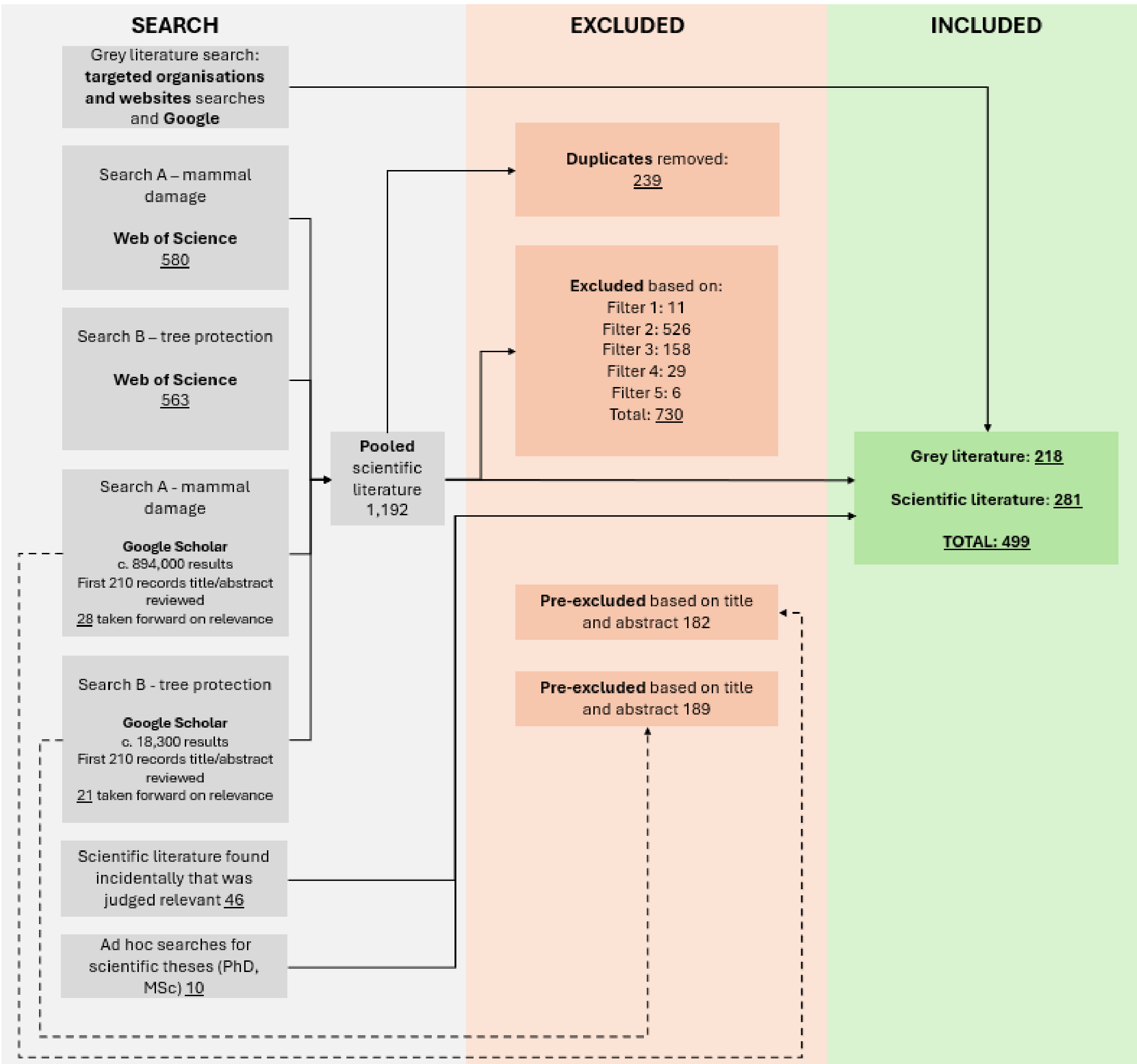

Figure 4. PRISMA flow chart of searches, exclusions and inclusions.

Precision of a systematic review can be defined as the number of included studies divided by the number of unique retrieved studies. In medical systematic reviews this can typically average 3% (Sampson et al., 2011). For our pooled scientific searches, we have included 281 papers as relevant, which is a precision of c. 24% including duplicates or 29% excluding duplicates. The higher value is expected because our inclusion criteria did not have strict specifications of study design, for example. Since grey literature searches are targeted, precision cannot be calculated.

3.2 Grey literature sources

A total of 218 grey literature sources were found from a variety of organisational sources (Table 6). Note we also searched without finding items deemed relevant to our specific research questions the following sites: Ancient Tree Forum, Coillte (Ireland's semi-state forestry company), England's Community Forests, European Commission Joint Research Centre, European Environment Agency, Fern, Food and Agriculture Organisation of the United Nations, Forest Canopy Foundation, Forest Ecosystem Restoration Initiative, Forestry England, Fountains Forestry Management Company, Global Partnership on Forest and Landscape Restoration, Joint Nature Conservation Committee (JNCC), Natural Resources Wales, Northern Ireland Department of Agriculture, Environment and Rural Affairs, People's Trust for Endangered Species, ProSilva, Society for Ecological Restoration, The Wildlife Trusts, Tilhill Forest Management Company, and United Nations Environment Programme. It should be noted that we only searched public facing websites rather than contacting organisations for literature.

Table 6. Breakdown of grey literature sources. These represent the first organisation we found the resource through, and some sources may be multi-authored and may be available on other sites.

Organisation/website	Number of sources found
Forest Research	61
Woodland Trust	16
NatureScot (and Scottish Natural Heritage)	14
Forestry Commission	12
How to Rewild	10
Forestry and Land Scotland	11
UK Squirrel Accord	11
Deer Initiative	10
Forest Plastics Working Group	8
Royal Forestry Society	7
British Wildlife Magazine	6
European Squirrel Initiative	6
British Association for Shooting and Conservation	5
Chartered Institute for Ecology and Environmental Management	5
Confor	4
Conservation Evidence (including journal)	4
Wild Deer Resource Scotland	4
Yorkshire Dales Millenium Trust	4
Gov.uk	3
The National Forest	3
Tubex	2
Game and Wildlife Conservation Trust	2
Gov.uk / DEFRA	2
Institute for Terrestrial Ecology	2

Institute for Chartered Foresters	2
Scottish Forestry	2
Sylva Foundation	2
Vincent Wildlife Trust	2
Woodlands of Ireland	2
Arboricultural Association	1
Centre for Ecology and Hydrology	1
Coed Cymru (Welsh Woodlands and Timber)	1
Department of Agriculture, Environment and Rural Affairs (Ireland)	1
Environment Agency	1
European Forest Institute	1
Farming Today	1
Natural England (and English Nature)	1
NERC Open Research Archive	1

3.3 Summary of articles by year, type, geography, taxa and damage

The dates of publications of relevant literature found in the review grew rapidly and in more recent sources (2020 onwards) grey literature was more common than scientific (Figure 5). The earliest scientific study found was from 1926 (Watt, 1926) and the earliest grey literature from 1958 (Rogers Brambell, 1958).

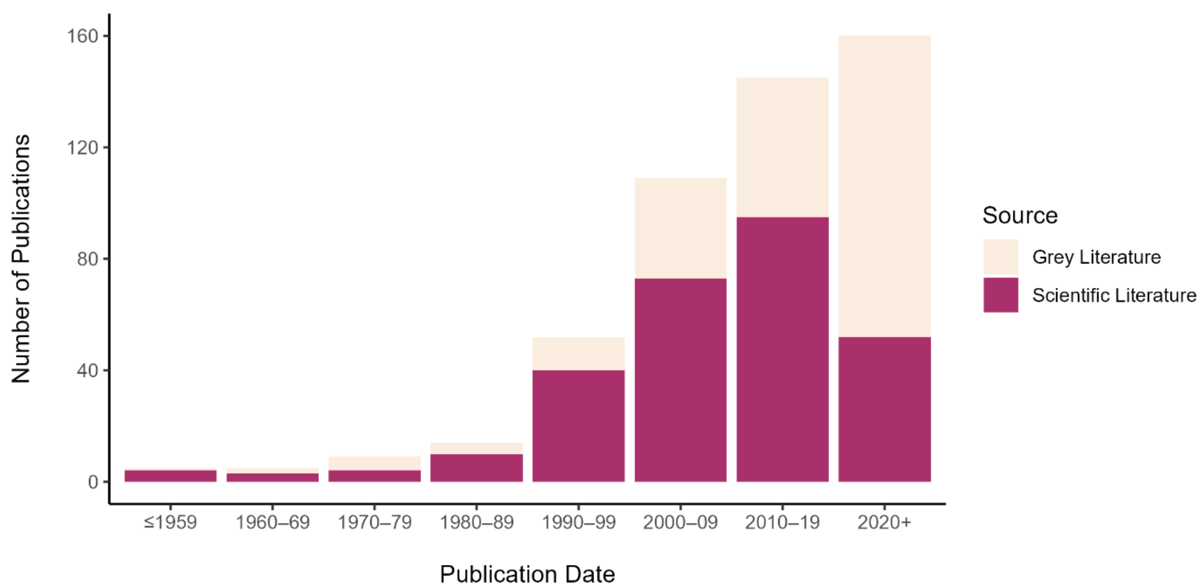

Figure 5. Trend in publication date by decade for scientific and grey literature identified in the systematic review.

Because our search and filtering were focussed on studies that referred to or are relevant to a UK geographic or taxonomic context, this resulted in most studies coming from the UK, but there is also good representation from countries in central, northern, and western Europe (Figure 6).

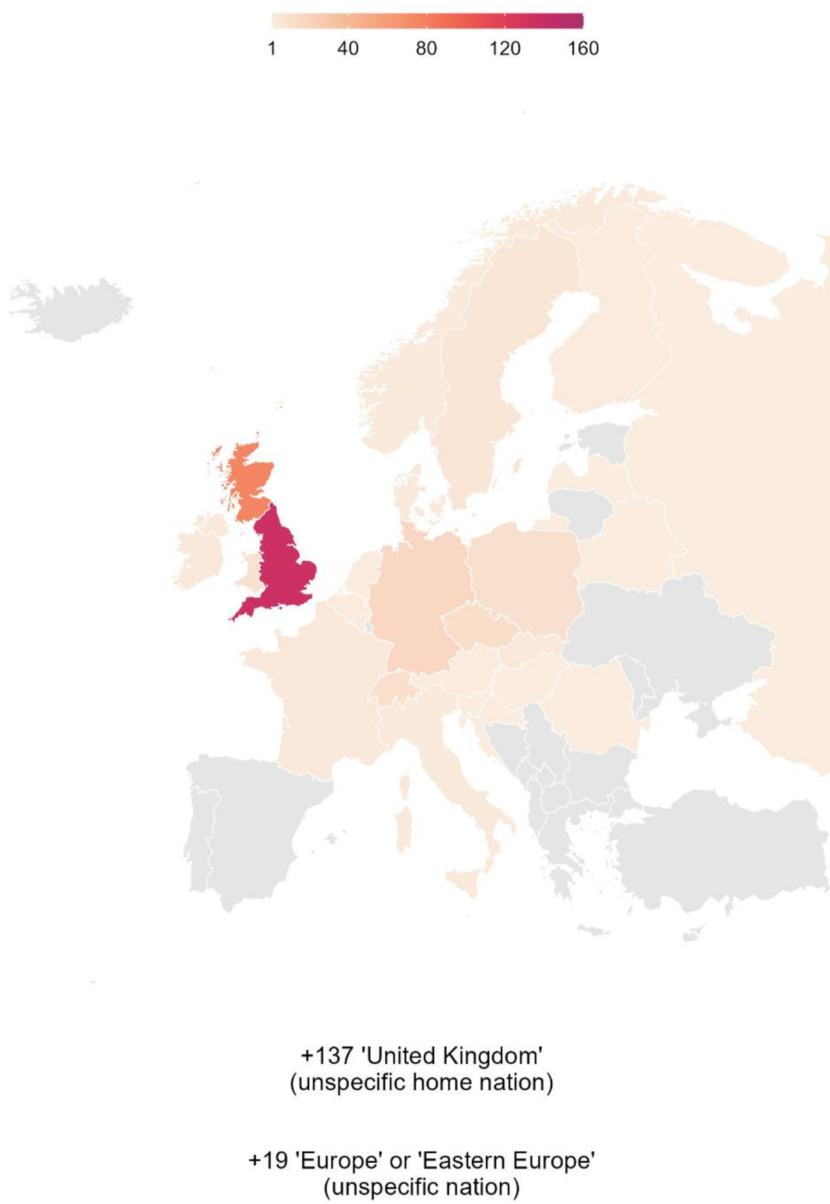

Figure 6. Geographic representation within sources found. Grey and scientific sources are pooled.

From the scientific sources found, a wide range of taxa are represented (Table 7) In terms of the literature itself, taxonomic groupings were not necessarily aligned perfectly to functional damage caused to trees. For example, beavers, voles, mice and grey squirrels are all Rodentia, yet functionally, damage they cause (and protection methods deployed) fit better into ‘Beaver’, ‘Small mammals’ and ‘Grey squirrel’ functional groups. We called such groupings ‘Damage functional groups’ and the synthesis of the literature review is grouped and ordered using this grouping (column 1 in Table 7).

Table 7. Frequency of reference to different mammal taxa within scientific sources found. Species are ordered taxonomically by Order and Family but also grouped into ‘Damage functional groups’ (first column) based on the types of damage and protection methods that frequently apply to that group.

Damage functional group	Order	Family	Genus/species	Number of sources		
Beaver	Rodentia	Castoridae	Eurasian beaver *Castor fiber*	14		
Small mammals		Cricetidae	Field vole *Microtus agrestris*	17	‘small mammals’ only = 8	
			Bank vole *Myodes glareolus*	24		
		Muridae	Yellow-necked mouse *Apodemus flavicollis*	7		
			Wood mouse *Apodemus sylvaticus*	15		
		Gliridae	Edible dormouse *Glis glis*	9		
Grey squirrel		Scirudiae	Grey squirrel *Sciurus carolinensis*	102		
Lagomorphs	Lagomorpha	Leproidae	European hare *Lepus europaeus*	35	‘lagomorphs’ only = 5	
			Mountain hare *Lepus timidus*	12		
			European rabbit *Oryctolagus cuniculus*	69		
Domestic pigs and wild boar	Artiodactyla	Suidae	Wild boar *Sus scrofa*	42		‘mixed large herbivores’ = 54
			Domestic pig *Sus domesticus*	23		
Deer		Cervidae	Roe deer *Capreolus capreolus*	113	‘deer’ (mixed / unspecified species) = 164	
			Red deer *Cervus elaphus*	100		
			Sika deer *Cervus nippon*	38		
			Fallow deer *Dama dama*	65		
			Reeve’s muntjac *Muntiacus reevesi*	52		
Domestic livestock and European bison		Bovidae	European bison *Bison bonasus*	8		
			Domestic cattle *Bos taurus*	74		
			Domestic/feral sheep *Ovis aries*	76		
			Domestic/feral goat *Capra hircus*	29		
Equines	Perissodactyla	Equidae	Domestic/feral horses/ponies *Equus ferus caballus*	52		

Mammal damage and resultant tree protection approaches are not always simple to classify. Species may have multiple impacts and different protection approaches are described in different ways or overlap in approach. In addition, browsing damage has difference facets and distinctions, such as in a commercial context where the extent of leader browsing may be crucial. Forms of damage can be related, for example browsing damage can lead to deformed tree growth. These aspects are discussed in more detail in the functional group accounts below. Nevertheless, types of damage recorded within studies are shown in Table 8.

Table 8. Frequency of different types of mammal damage referred to within scientific sources that we found. This gives a broad overview of how mammals can impact tree health, growth or survival, and demonstrates which types of damage appear most often in the literature. Note, types of damage may be referenced in multiple sources, reflecting a cross-reference of impacts that are not mutually exclusive.

Type of damage	Number of sources
Browsing (foliage, buds)	195
Bark stripping	133
Preventing regeneration/colonisation	111
Unspecified herbivory	46
Fraying	44
Seedling herbivory	39
Deformed tree growth (including forking)	36
Seed predation	33
Economic damage	30
Trampling of seedlings	21
Rot from mammal-induced wounds	19
Breaking stems	16
Tree felling	14
Rooting	14
Damage fences/ditches/banks	13
Uprooting	12
Girdling (due to excessive bark stripping)	11
Staining on processed timber	10
Disease transmission	6
Flooding of commercial stock	6
Bole scoring	4
Soil compaction	4
Promote windthrow	4
Restrict access for other forestry management activities	4
Reduce flowering	3
Dung and urine damaging mycorrhiza	1

3.4 An ecological framework for tree protection approaches

There is a very broad array of tree protection methods described in the literature. Furthermore, the types of approaches that have been used, tested or suggested for different species vary widely. We can foresee the array of different options for practitioners being very large. This will present a challenge in terms of decision-making based on species, management objective and other constraints such as budget and scale. Based on our reading across the literature we developed a framework for classifying tree protection methods in an ecological sense. This considers mammals as ‘consumers’ and trees as ‘resources’. Damage to trees is thus resource consumption (although note in some cases such as ‘fraying’ by deer, the tree is providing an alternative resource in the form of velvet removal or territory marking).

Under this framework, from the literature we identified that managers can either (1) monitor damage or assess risk prior to any intervention or, if damage or risk of damage is deemed unacceptable given management objectives, can take action by (2) directly reduce mammal numbers or their demand for feeding on trees, (3) protecting trees or groups of trees via physical or sensory barriers, or (4) modifying woodland management approach to reduce vulnerability. Most tree protection methods we identified functioned through one of these broad approaches, and this framework is outlined in Figure 7. Broadly, the options within 2, 3 and 4 sit along a spectrum of management focus as follows:

- management of mammals themselves
- physical barriers or deterrent approaches (keeping mammals and trees apart)
- managing trees or altering woodland management to reduce demand by mammals or increase resilience to damage

Because beavers can cause indirect impacts on trees via flooding, a separate set of methods for mitigating this risk is included for this species only.

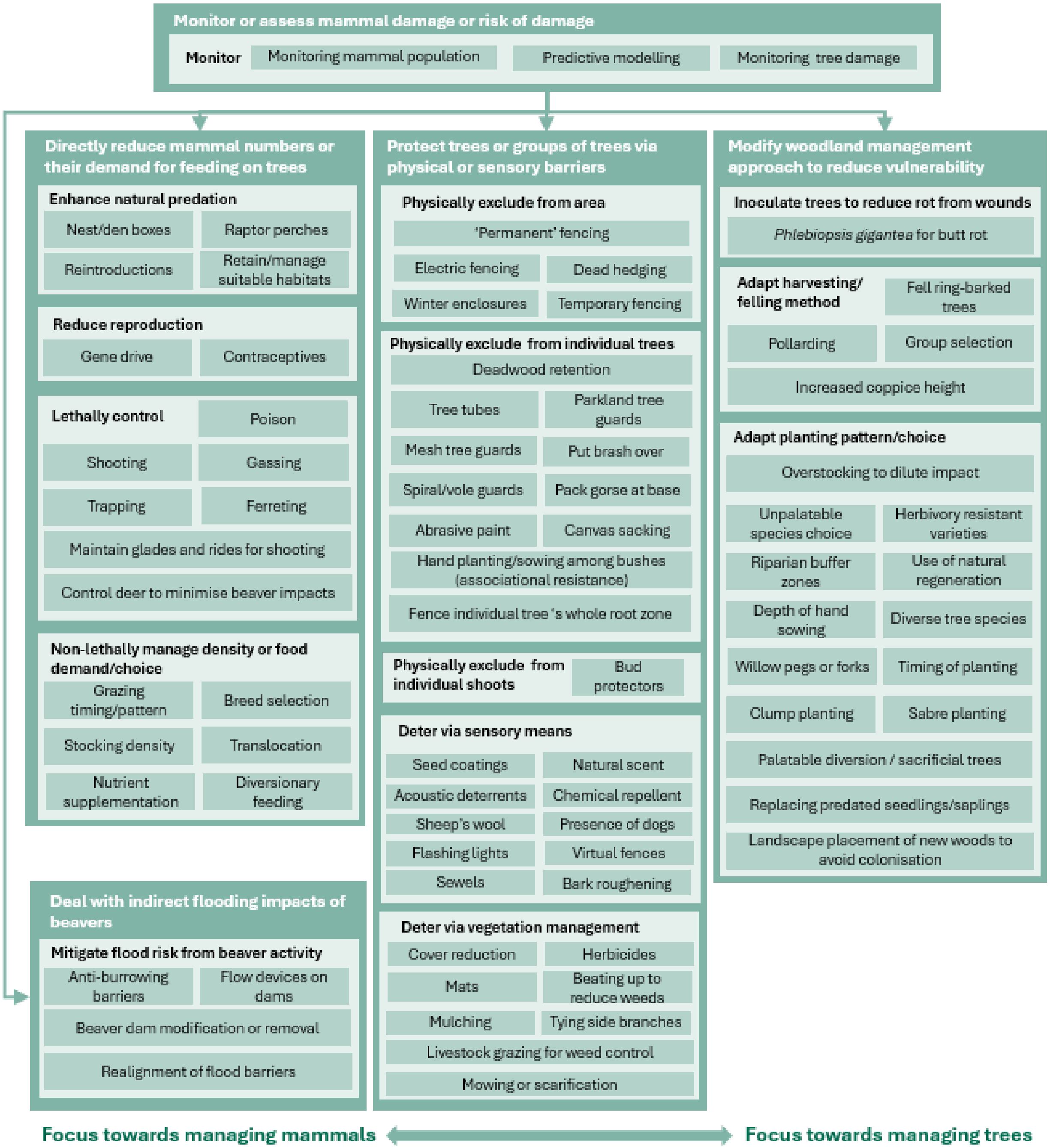

Figure 7. Ecological framework for categorising tree protection methods. Assessing damage or risk of damage itself forms a protection method and may be the first step. However, if, given management objectives and resources tree protection is required, then methods can be adopted that either reduce tree resource demand by mammals. modify access to those tree resources, or modify tree resource availability or resilience to damage.

3.4.1 Applying the framework to mammal functional groups

In this section we apply this framework to each damage functional group in turn, identifying which tree protection methods appear in the literature for which groups, although recognising which options are feasible or desirable, may also depend on the management objectives and other considerations. Because for some methods there was more extensive scientific evidence but for others the evidence base was more anecdotal or less strong, we adopted a simple traffic light system for each group: (i) not used or no evidence base for that method for that mammal group, (ii) weaker evidence base or occasional use noted in literature, and (iii) stronger evidence based or widespread use evident in literature. This approach provides a visual representation of the specific methods potentially available for a mammal group as well as where they fall within the framework in Figure 7. Maps of each of the eight species or functional group against each protection method identified are given in Figures 8 to 15.

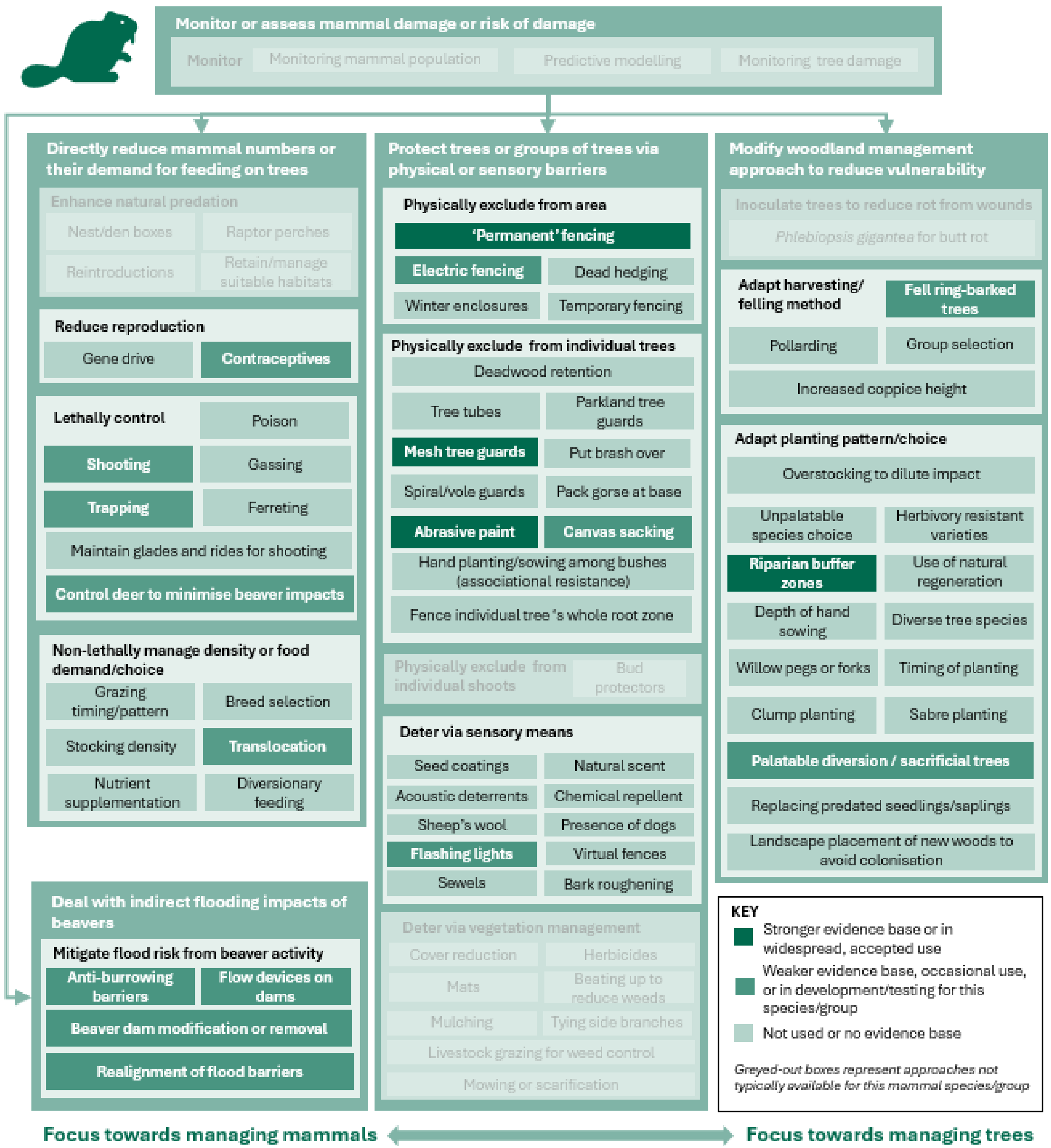

Figure 8. Map of tree protection methods identified in the literature for beavers (family: Castoridae) against tree protection framework.

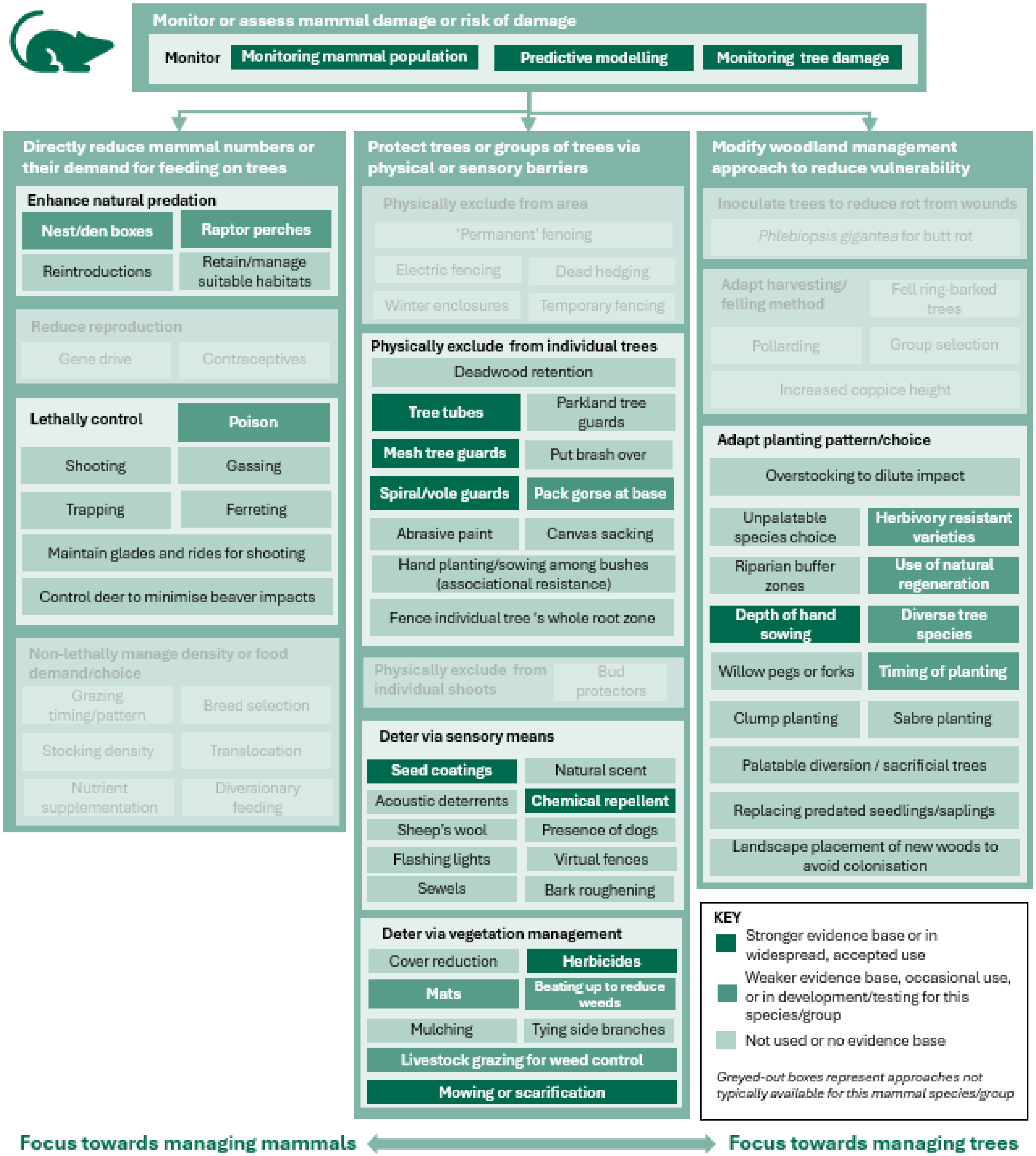

Figure 9. Map of tree protection methods identified in the literature for small mammals (families: Cricetidae, Muridae, and Gliridae) against tree protection framework.

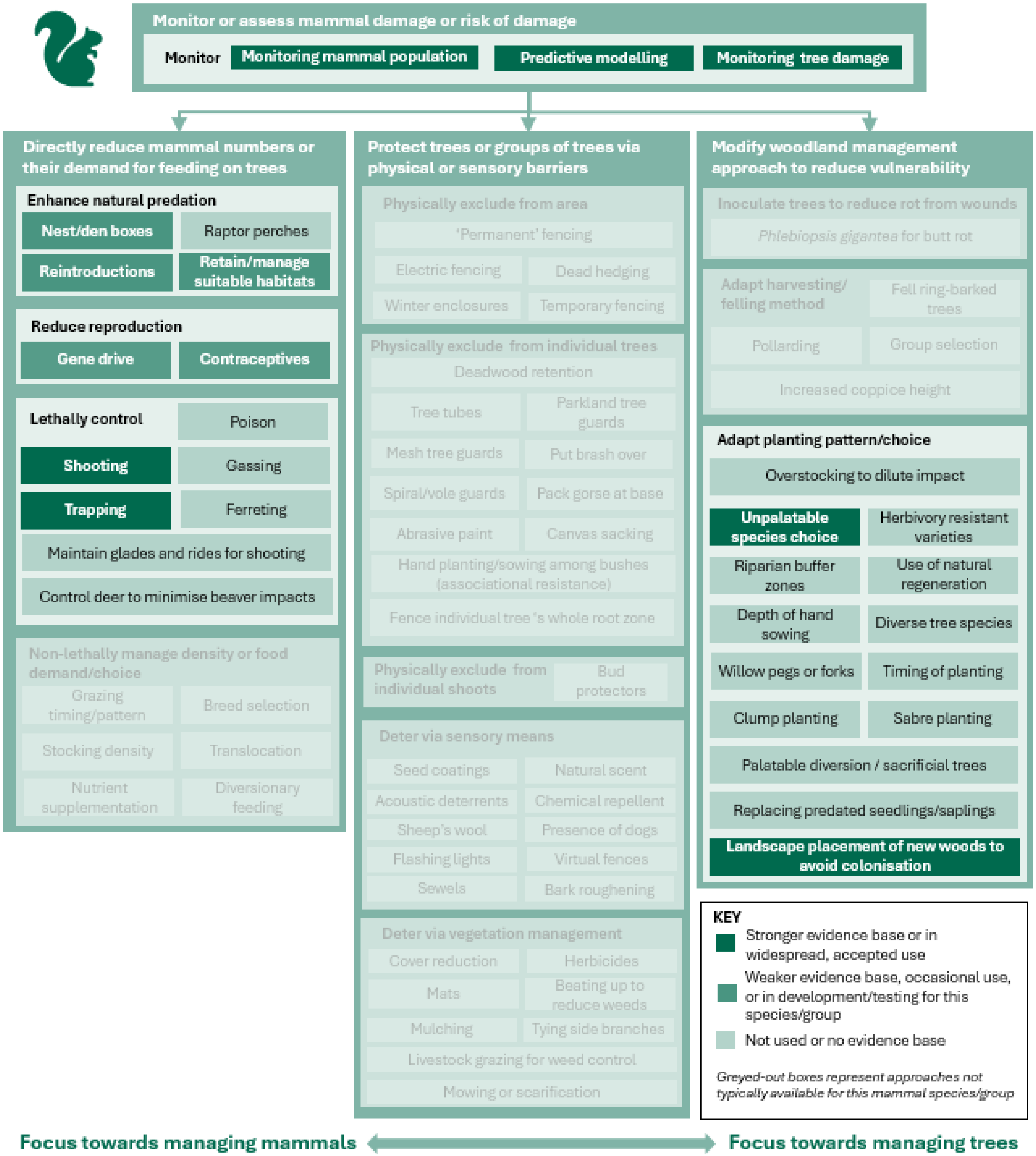

Figure 10. Map of tree protection methods identified in the literature for grey squirrels (family: Sciuridae) against tree protection framework.

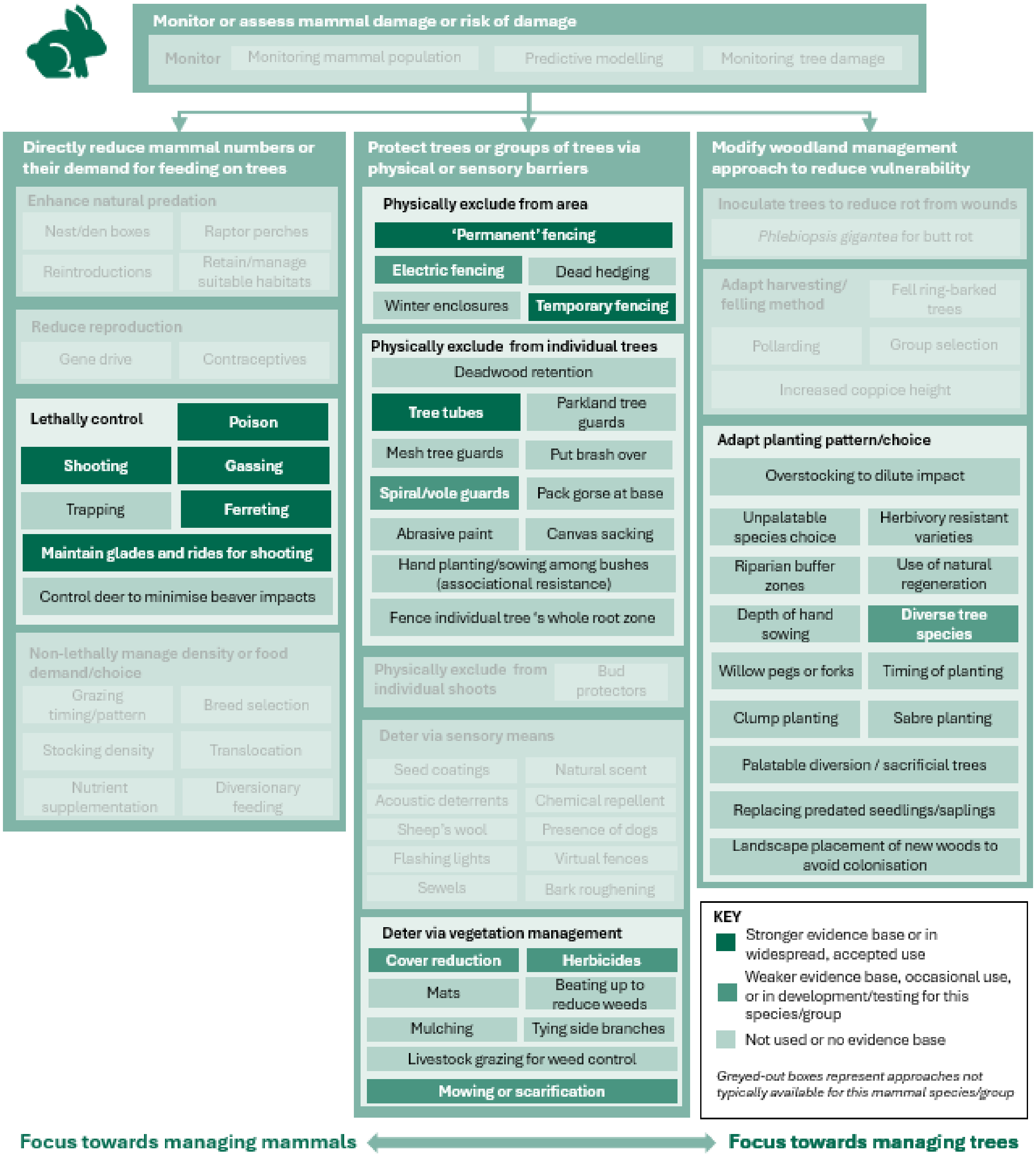

Figure 11. Map of tree protection methods identified in the literature for lagomorphs (i.e., rabbits and hares) (family: Leporidae) against tree protection framework.

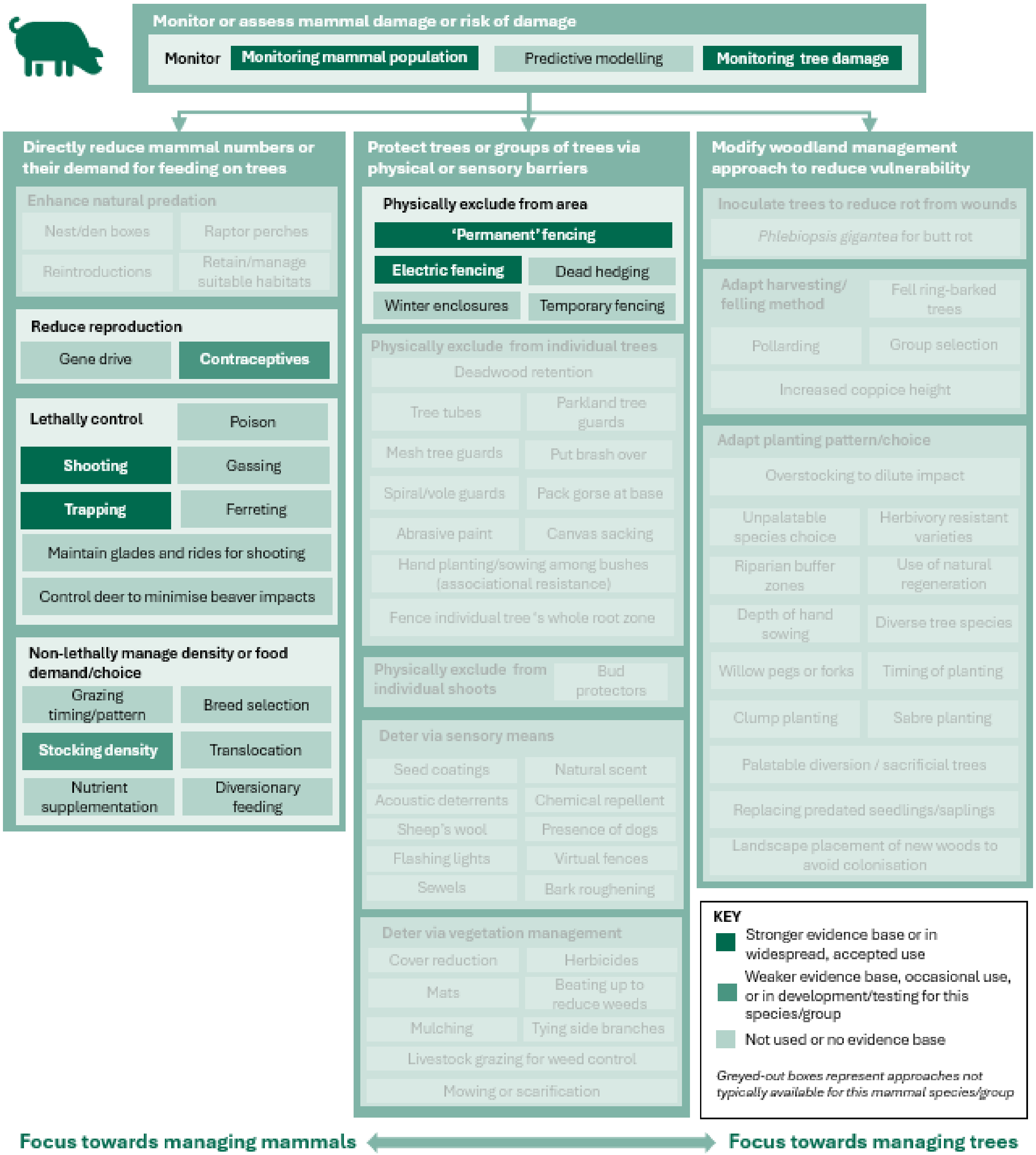

Figure 12. Map of tree protection methods identified in the literature for pigs and wild boar (family: Suidae) against tree protection framework.

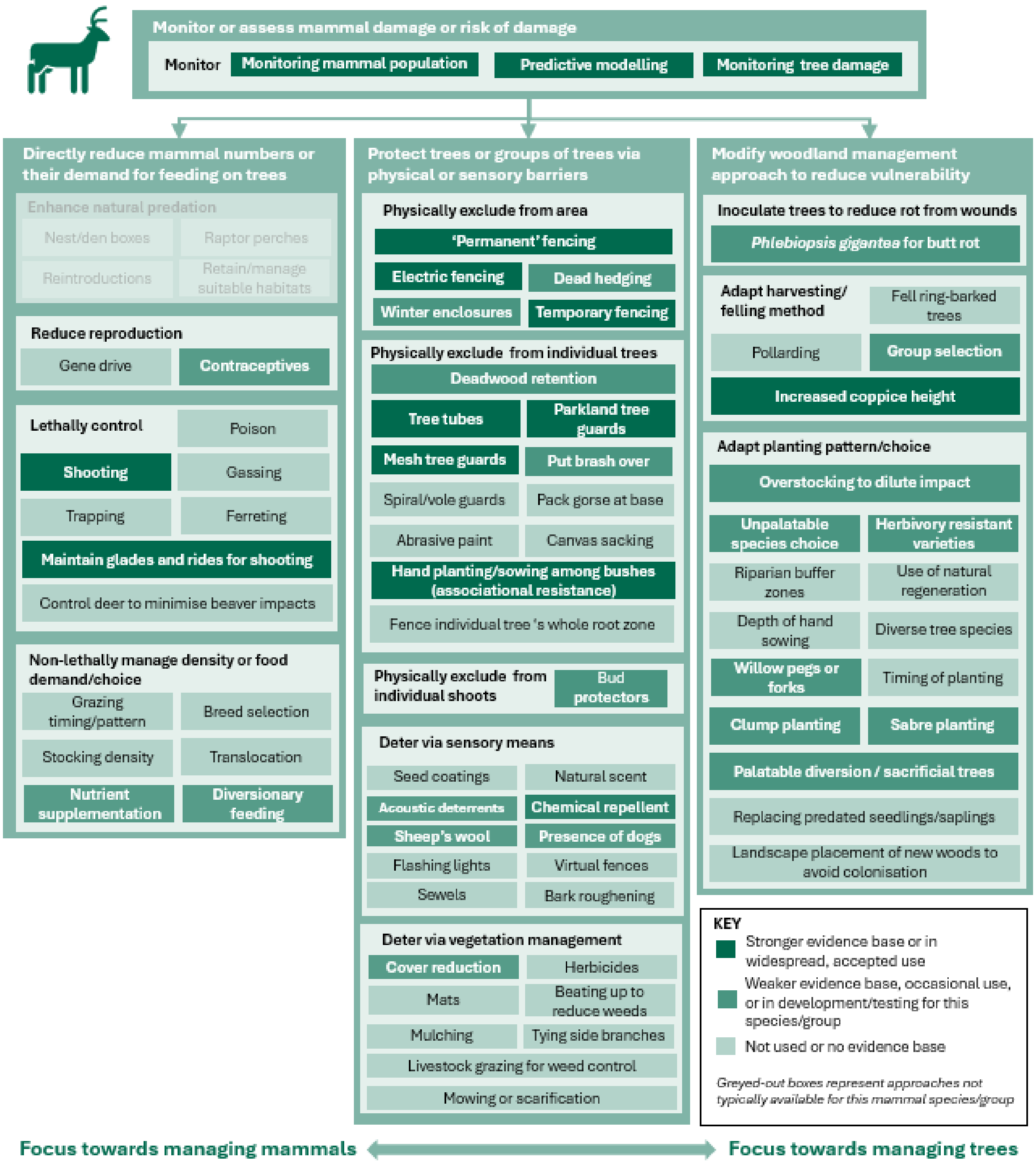

Figure 13. Map of tree protection methods identified in the literature for deer (family: Cervidae) against tree protection framework.

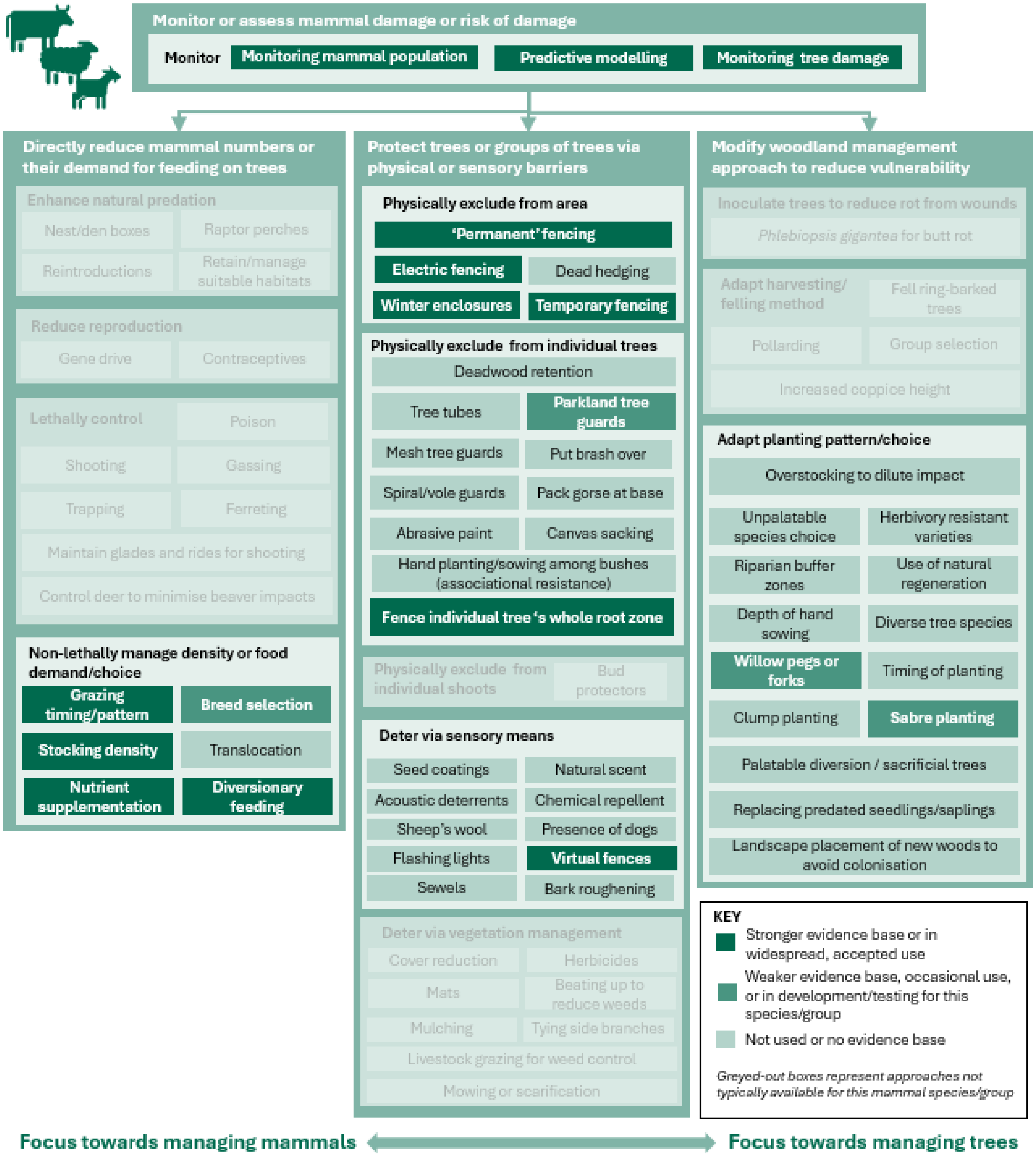

Figure 14. Map of tree protection methods identified in the literature for livestock (i.e., sheep, goats, cattle, and bison) against tree protection framework.

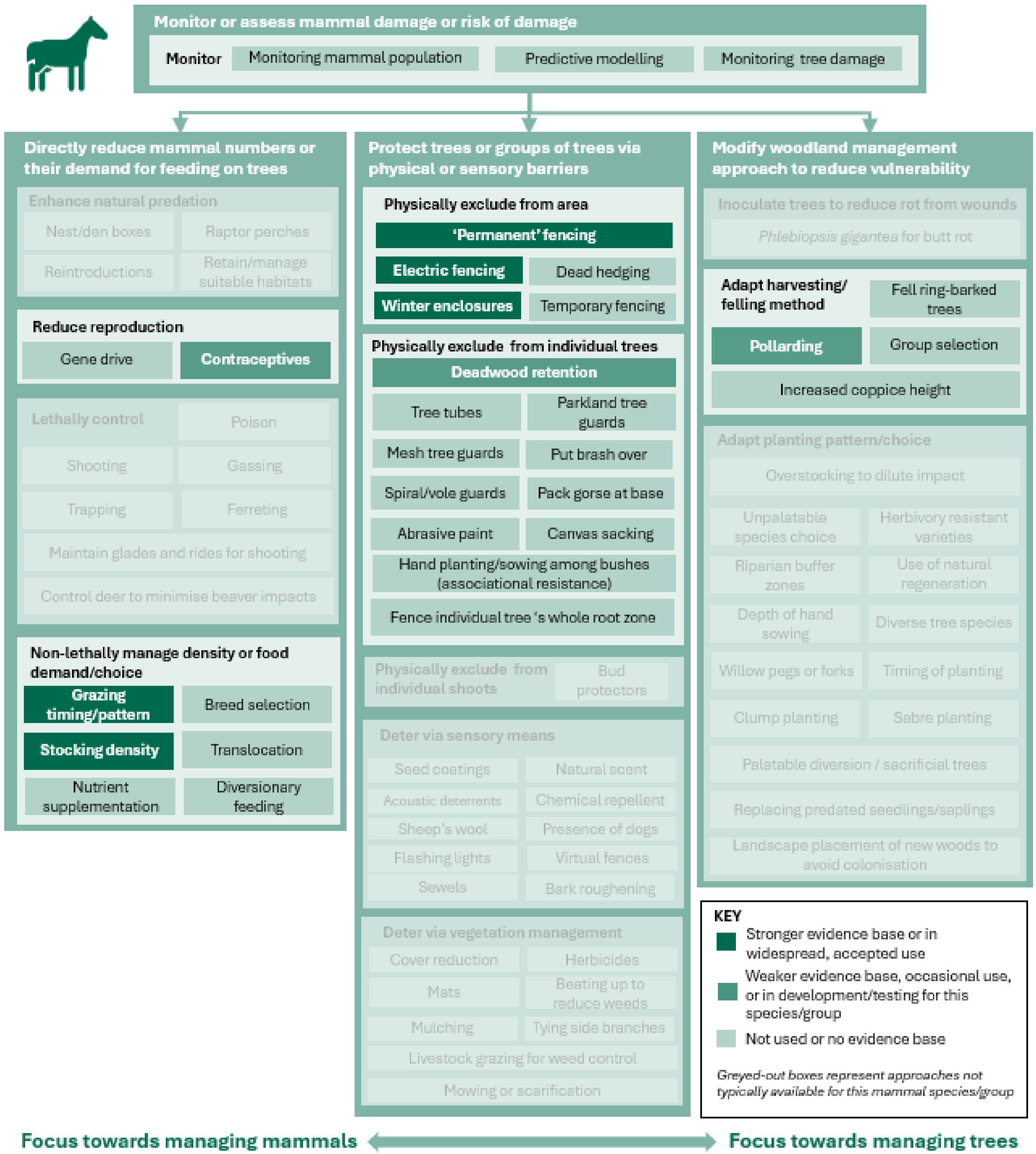

Figure 15. Map of tree protection methods identified in the literature for horses and ponies (family: Equidae) against tree protection framework.

4 DISCUSSION

It was evident across the review that for each species or functional group (Figures 8-11) there are a range of potential protection methods, but where they fit in the framework and their quantity differ widely between the groups. The reason for differences between protection options available for these groups is a likely a function of aspects such as the group/species' ecology, behaviour, type of damage caused and legal or logistical restrictions around certain approaches. For some groups like deer (Figure 13) a very wide array of approaches have been tested or suggested, that cover all of the different ecological intervention points (i.e. monitoring, focussing on mammal management or focussing on tree management approaches). This perhaps reflects the substantial impact deer species have had on trees and woodland in England and the wider region, providing an economic driver for development of protection approaches. In some cases, certain protection approaches are not viable for a given functional group or species. For example, for grey squirrel (Figure 10) exclusion approaches (fencing, individual tree protection or deterrents) are largely unfeasible due to the species' behaviour and the tree life stage it primarily impacts. For other groups an approach that may have some effectiveness for one species, may be ineffective against others. For example, associational resistance (the phenomenon whereby a plant species is less susceptible to herbivory if found associated with other plant species that reduce herbivore access to it, such as an oak tree growing among thorny shrubs) may be effective against deer, but unlikely to work with rabbits and hares (Figure 11) who can access dense and/or thorny scrub.

4.1 Unintended consequences of tree protection methods

We focus on the details of evidence and context for these protection methods by group in the Appendix, below. One thing that we noticed when reviewing the literature was the requirement in the technical guidance document to consider nuance and potential unintended consequences of tree protection approaches, and interactions between species. For example, stock fencing has been seen to increase rank ground flora inside enclosures which can then encourage rabbits (reported in: Thomas et al., 2007) or field voles (Andrews et al., 2000). Furthermore, exclusion of rabbits may also increase ground vegetation biomass which might increase vole populations (Trout et al., 2004). In terms of ecosystem services, where woodland is fenced to exclude large herbivores, the opportunity for them to disperse seeds to neighbouring rewilding projects is limited (Spencer, 2023). Temporary dead hedging for exclusion of deer could provide cover for rabbits if present (H. Armstrong et al., 2003). Similarly, as discussed in the synthesis in the appendices, in an experimental study in Germany, while the retention of deadwood in the form of lying tree crowns reduced deer browsing probability on silver firs saplings, rodent browsing (identified as most likely bank vole and yellow-necked mouse) increased with more lying deadwood (Hagge et al., 2019), perhaps as a result of increased cover.

Supplementary feeding of livestock to discourage tree damage requires feed to be brought in by vehicle, with associated tracks and ruts and perhaps packaging litter (Thompson et al., 2005), may concentrate trampling (Scottish Forestry, 2024) and potentially increase disease transmission via aggregation of individuals (Mysterud et al., 2023). In the Alps where the aim was to attract Highland cattle to suppress green alder (*Alnus viridis*) growth, molasses blocks increased browsing of the area within 10m of the blocks compared to the previous year and to supplemented areas, with a corresponding increase in bare soil and decrease in cover of ferns, herbs and woody plants (Svensk et al., 2022). Other examples include some evidence that trees grown without tree tubes can be more resilient to weather extremes and drought (Coutand, Dupraz, Jaouen, Ploquin, & Adam, 2008 cited in: Woodland Trust, 2022), and provision of nest boxes for raptors may decrease local populations of passerine birds (Paz Luna et al., 2020).

4.2 Considering carbon costs of tree protection methods

We found some limited evidence in the literature about carbon costs of tree protection methods. Life-cycle assessment of the carbon costs of tree protection (Chau et al., 2021a, 2021b) show that beating up (periodically replacing lost saplings) has carbon emissions about three or four times lower than scenarios using tree shelters, despite the additional sapling production and transport needed. This calculation assumed 85% survival with tree shelters and holds true unless survival without tree shelters declines below 30%. If tree shelters were collected after five years for recycling, survival without shelters need only decline to 38% to tip the balance in favour of using shelters. The viability of this scenario is uncertain since many tree shelters are not currently collected, and molecular degradation of plastic shelters means that part-recycling or downcycling is more likely. Biodegradable shelters of polylactic acid or bio-polypropylene resins require more resources and energy to manufacture than standard polypropylene and take a long time to biodegrade. As such, current bioplastic options are not preferable to standard plastic shelters, at least in carbon terms. Analyses based on greenhouse gas emissions do not account for other ecological impacts: of the scenarios analysed polylactic acid shelters left to degrade in situ would generate the greatest number of plastic fragments; this material also performed worst across a panel of other pollution categories (Chau et al., 2021b). For all protection methods, the carbon storage of a surviving tree in its first 25 years of growth far exceeds the emissions associated with manufacture, transport and degradation of plastic tree shelters (Chau et al., 2021a, 2021b), although even for future scenarios in which tree shelters are fully recoverable and recyclable, establishing a tree without a shelter may still have less than half the carbon emissions of a fully recyclable shelter. In respect of other impacts such as resource use and pollution the trade-offs are complex. However, planting without shelters generates the lowest impacts in seven and the highest impacts in four (due to the requirement for watering in nurseries and copper oxide fungicides) of the 15 categories examined and is therefore the preferable option (Chau et al., 2021b).

4.3 Format of literature

The vast majority of the literature identified was in the form of PDF documents or single text-focussed websites, although some such as the Wild Deer Best Practice Guidance for Scotland (NatureScot, 2024) is in the form of an illustrated website. Many of these do contain engaging illustrations relating to tree damage or protection methods, which are tagged in the resource bank for later access. Today the web and smart phones offer more opportunities for immediate access to guidance and in multimedia formats. Webinars and other video media offer easily accessible information (e.g. UK Squirrel Accord, 2022). Increasingly apps are being developed to aid collection and collation of wildlife management data that can be used in the field (e.g. Sylva Foundation, 2024; WildTrackPro, 2024).

4.4 Synthesis of damage and protection approaches for functional groups

In the Appendices, we synthesise information on each functional group or species (Table 7) in turn. This includes elements from the literature relating to the ecosystem services of that taxa, the damage/impacts it has on trees and treescapes, identification of that damage relative to other taxa, and tree protection methods within the framework of Figure 7.

5 APPENDICES

5.1 Notes on the appendices

Where we report any costs relating to damage or protection methods, we translate historical costs at time of publication of the source (or the data if stated) in approximate costs today. These estimates are only approximate and based on the Bank of England Inflation Calculator (https://www.bankofengland.co.uk/monetary-policy/inflation/inflation-calculator). Wild taxa are treated in taxonomic order and taxa that are domesticated or include domesticated individuals (e.g. goats, ponies) are at the end. Note that where some species' geographical ranges are referred to, they come from IUCN Red List (https://www.iucnredlist.org/) or Matthews et al. (2018), and presented in the taxonomic order of the latter. To aid navigation and for providing information for the technical guidance document, sections have been laid out in a standard form as per Box 7. Protection methods have been underlined in the text for ease of scanning for different approaches per group/species.

Box 7. Structure of each functional group or species account in the below appendices. The section relating to beavers has a couple of unique section headers due to the unique nature of their impacts.

Background and ecosystem services
Damage and impacts
Identification of damage
Protection against damage

- ***MONITORING BEFORE INTERVENTION***
- ***REDUCE TREE RESOURCE DEMAND***
- ***MODIFY ACCESS TO TREE RESOURCES BUT NOT AVAILABILITY***
- ***MODIFY TREE RESOURCE AVAILABILITY AND RESILIENCE TO DAMAGE***

5.1.1 Beaver (Family Castoridae)

5.1.1.1 Background and ecosystem services

Of all species considered here, beavers are perhaps best known for their influence on ecosystems, as so-called ecosystem engineers (How to Rewild, 2024a). Beavers are likely to have become extinct in Britain in the 14th Century, but following illegal and later legal trial reintroductions in the early 21st century they are now well-established in some areas, particularly Tayside in Scotland, and have since been recategorized as a native species in Scotland (The Scottish Government, 2017) and subsequently England (Bird-Halton and Natural England, 2022). In England, National

Biodiversity Network data (https://records.nbnatlas.org/) since 2020 show concentrations of records of beavers in the Southwest, West Midlands and Kent, with several other isolated records.

Potential ecosystem benefits of beavers include benefits to natural flood management (Environment Agency, 2024), creation of more diverse woodland structure in terms of age class and volume of standing deadwood, and benefits to diversity of various other taxa via creation of ponds and wetlands, deadwood and more diverse woodland structure. In the case of the latter this may depend on context such as initial woodland structure and presence of other herbivores (The Scottish Government, 2017). Macdonald et al. (1995) list potential positive impacts of beavers as including improved water quality, preventing downstream flooding, creating wetlands, and raising water tables, and some negative aspects including flooding woodland, feeding on softwoods and burrowing through dykes. Generally, it is predicted that beavers should create a younger age profile in a woodland because younger trees produce more and stronger regrowth shoots following browsing (The Scottish Government, 2017). In addition, as central-place foragers, i.e. foraging from a central place of refuge (the lodge), then it is predicted that their impacts will create special heterogeneity in woodland structure. The overall effect of beavers over four years of the Scottish Beaver Trial was to create a more open woodland canopy, with lower vertical density, and a ground layer with more grass and woody debris, and less leaf litter (Iason et al., 2014).

Stringer and Gaywood (2016) carried out a metanalysis of positive and negative impacts plants, invertebrates, amphibians, reptiles, birds and other mammals. They found generally positive impacts on other elements of biodiversity, most likely via impact on spatial and structural habitat heterogeneity (including creation of ecotones) or creation of unique habitats.

5.1.1.2 Damage and impacts

It has been noted that the apparent impact of beavers will be exacerbated where deer densities are high, because regrowth and regeneration following beaver impacts will be hampered (Gaywood, 2015; Stringer and Gaywood, 2016). Beavers specifically select area with woodland for their territories, needing suitable woodland or scrub within 50 m of freshwater and about 2 km of woodland for each 4 km stretch of river bank (The Scottish Government, 2017). Although considered a potential pest to agricultural land, in terms of forests and forestry, it is both the direct impact on trees via browsing and felling, and indirect impact of flooding that might be considered as aspects of tree damage (How to Rewild, 2024a). Felling occurs both for feeding and for dam construction (The Scottish Government, 2017) Beaver dams are known to benefits so-called ‘natural flood management’, storing and slowing water (Environment Agency, 2024). Thus, the impact of beavers on flooding appears to be a trade-off between local flooding, which will occur behind dams, and the reduction in flooding severity downstream. In terms of overall impact, Gaywood (2015) reports surveys which suggested that 0.01% of agricultural and forestry land in Poland and 0.1% of forestry land in an area of Norway were affected by flooding from beaver activity, although at a local level any impact could be important.

Some species are relatively tolerant to partial or seasonal flooding and so may be favoured by beaver presence, such as willow, while others are not tolerant, such as elm or juniper (Campbell-Palmer et al., 2015).

In terms of selection of trees for feeding, they generally avoid conifer species, but can still impact conifers via flooding, most species of which are not tolerant of prolonged flooding (The Scottish Government, 2017). It has been noted, however, that beavers may occasionally ringbark conifers or feed on conifer saplings in late winter or early spring where few broadleaved trees are available (Gaywood, 2015; The Scottish Government, 2017). Flooding may also have an impact of forestry infrastructure and access (The Scottish Government, 2017). Gaywood (2015) crudely estimated that forestry losses in Scotland, if similar to those experience in Norway, could amount to about £400-500 ha at a cost of c. £3-7M . It is challenging to translate this to an English context due to differences in forest cover and type of commercial forestry.

While avoiding conifers, they appear to select for aspen and willow (The Scottish Government, 2017) and also ash and hazel (Iason et al., 2014). High amounts of beaver felling or damage to birch have been reported (Campbell-Palmer et al., 2015) but during the Scottish Beaver Trial birch was found to be impacted at a lower rate than expected based on its availability (Iason et al., 2014). As much UK-focussed impact assessment has occurred in Scotland where the principal commercial species are conifers, potential impacts on commercial broadleaves, more common in England, are less clear.

Although known for felling large trees, smaller stems (<10 cm diameter) are preferred where available. Selective browsing by beavers can reduce tree diversity and prevent regeneration and regrowth, particularly when in combination with deer or livestock browsing pressure (Stringer and Gaywood, 2016; The Scottish Government, 2017). In terms of the latter, natural coppice regrowth that can occur as a result of beaver felling can be completely removed by deer and/or livestock, and aspen can be at risk as it is selected for felling by beavers and attractive to deer.

Although The Scottish Government (2017) suggest beavers can benefit natural regeneration via canopy opening, Lyons et al. (2024) note that beavers can prevent succession via direct felling of trees and scrub, and in flooding caused by dams. In addition, where woodland is already fairly open, felling might lead to loss of canopy layer (The Scottish Government, 2017)

5.1.1.3 Identification of damage

Beaver damage is almost unmistakable, but useful pictures are provided in Campbell-Palmer et al. (2016) including teeth marks, felled and gnawed trees and bark-stripping (including ring barking) .

5.1.1.4 Protection against damage

Tree protection methods identified in the literature for beavers are shown in Figure 8, with details and evidence base synthesised below. As beavers are a European Protected Species, Gaywood (2015) identified the actions with beavers that would or would not require derogation licenses under the Habitats Regulations in Scotland, and it seems likely similar will apply in England.

MITIGATION OF EFFECTS OF DAMS OR BURROWING

Because unlike any other species in this review beavers can create flooding, here we discuss indirect tree protection methods that are specific to preventing dams from creating too much flooding or burrowing into flood banks. An approach adopted is <u>using flow devices to bypass dams</u> to avoid flooding where this is likely to impact forestry or agriculture (Gaywood, 2015; How to Rewild, 2024a). These can enable continued presence of dams/lodges while managing water level to a degree considered acceptable. Approaches such as, <u>dam-notching</u> (removal of a small section of dam) or <u>dam removal</u> may reduce flooding in the short term but can be repaired or replaced by beavers rapidly and they may stimulate beavers to cut fresh woody material for rebuilding (Campbell-Palmer et al., 2015; Gaywood, 2015; The Scottish Government, 2017). It should also be noted that destruction of a dam associated with a lodge would likely require a derogation license under the Habitats regulations, as in Scotland (Gaywood, 2015).

Where burrowing into flood banks is an issue or risk, <u>anti-burrowing approaches</u> used in other burrowing species such as insertion of sheet metal piling or welded wire fabric into flood banks, or using rocks, concrete or stone gabions on the water-facing bank have been reported to have been effective, while geotextiles have not (Campbell-Palmer et al., 2015). It has been noted that the wider ecological consequences of such structures is important to consider (Campbell-Palmer et al., 2015). <u>Realignment of flood banks</u> so that there is 10-20 m strip of bankside vegetation or wet woodland within the flood bank has been suggested as potentially effective, if expensive (Campbell-Palmer et al., 2015).

MANAGEMENT OF OTHER MAMMAL SPECIES WHERE BEAVERS ARE PRESENT

Beaver cut or felled trees such as willow and aspen have good capacity to resprout and in a study in a 2-year study in a captive enclosure, beavers did not harvest regrowth (Jones et al., 2009). However, because beavers are highly protected and their impacts are likely to be quite localised to near waterbodies, one management tool for tree protection in beaver-occupied areas suggested has been to reduce the cumulative impacts of beavers and other herbivores, by controlling damage by the other herbivores rather than beavers. In the Scottish Beaver trial, two-thirds of resprouted shoots of beaver felled-trees were browsed by deer (Iason et al., 2014). One mitigation measure against tree damage by beavers, particularly low regeneration or retarded coppice regrowth is via <u>management of impacts other herbivores</u> such as deer or livestock via fencing or individual tree protection (The Scottish Government, 2017). Stringer and Gaywood (2016) suggest a coordinated approach to deer and beaver management is required.

REDUCE TREE RESOURCE DEMAND

A range of approaches to discourage beavers dam building or setting up territories in an area have been tested, such as <u>electric fencing</u> or <u>flashing lights</u> but, as reported by Scottish Government (2017) and Gaywood (2015) these have been ineffective although Campbell-Palmer et al. (2015) report these can be ‘temporarily effective’. Campbell-Palmer et al. (2015) also report other idiosyncratic approaches such as barrels hanging on chains or compact discs suspended on cords as deterrents to prevent dam building or rebuilding following removal, and thus local establishment of territories.

Culling of beavers where no other suitable alternative is available would require a license, but is a beaver management approach used in several European countries where beaver densities and distributions are larger (Gaywood, 2015). Campbell-Palmer et al. (2015) note that trying to establish ‘beaver-free zones’ is likely to involve high cost and continued removal effort. Trapping for lethal control, relocation or sterilisation have also been discussed for beavers, as has darting with contraceptive drugs (Gaywood, 2015). These options might be considered where the continued costs of other options are expensive and time consuming. As beavers become more common and their conservation status improves, there may be more potential for licensing such activities (Gaywood, 2015). Detailed description of practicalities, effectiveness, and animal welfare issues around these activities are discussed in Campbell-Palmer et al. (2015)

MODIFY ACCESS TO TREE RESOURCES BUT NOT AVAILABILITY

Fencing can be an option for reducing beaver damage to trees although it requires suitable maintenance (The Scottish Government, 2017), and Gaywood (2015) notes that using it to prevent movement along waterways rather than access to trees may simply provide a dam-building point. Campbell-Palmer et al. (2015) suggest this could be used for individual stands of trees or to discourage colonisation of small streams or ditches. Permanent beaver fencing is likely to be most effective when set back from water courses (20-30 m), and should extend underground or have a mesh collar at ground level to discourage digging under (Campbell-Palmer et al., 2015). It should also be considered that beaver tree-felling makes fence damage more likely so managing trees near to fences or considering placement carefully is important (Campbell-Palmer et al., 2015). A type of fence called a ‘Swept-Wing Fence’ has been designed to cause beavers to double back on themselves when exploring up a small water course (Campbell-Palmer et al., 2015). Indicative costs of fencing are laid out by Campbell-Palmer et al. (2015). Temporary electric fencing has been used to prevent beaver access to arable crops (Campbell-Palmer et al., 2016), although these are available as a food source for a relatively short period of time, while tree resources within beaver territories may need longer-term protection.

An approach to reduce damage cause by beaver felling is individual tree protection, which can include the use of wire mesh. NatureScot (2020) provide technical specifications for required mesh size, wire thickness (thin ‘chicken wire’ is likely to be pulled down), and overall height (90 cm) and an instructional video demonstrating installation. It is noted that some spacing between the mesh and the tree is required to prevent bark gnawing through the mesh. Another approach mentioned by Scottish Government (2017) is the use of deterrent paints, and Campbell-Palmer et al. (2015) indicate some options such as an anti-game paint “Wobra” or a mix of non-toxic exterior paint and mason sand which was apparently effective in a trial in Scotland. Macdonald et al. (1995) also mention that canvas sacking 1 m high around the base of a tree can be effective.

Use of flashing lights, acoustic ultrasound deterrents or the presence of free running dogs have also been suggested as possible deterrents although without reporting of evidence for effectiveness (Campbell-Palmer et al., 2015).

MODIFY TREE RESOURCE AVAILABILITY AND RESILIENCE TO DAMAGE

The Scottish Government (2017) suggest that riparian buffer zones that are already in place to protect water courses are probably in most cases sufficient to reduce potential beaver feeding damage substantially in terms of commercial species. However, this may not necessarily mitigate the risk of flooding if this occurs beyond this zone. Where designing new woodland, leaving such buffer zones could prevent damage to trees in that new woodland, and values between 20 m and 50 m from watercourses have been suggested. During the Scottish beaver trial, over four years of monitoring the majority of effects of beaver foraging on trees remained withing 10 m of the water's edge (Iason et al., 2014). In addition, a form of sacrificial planting could be used, i.e. planting and management of native trees or encouraging natural regeneration in the riparian zone to produce suitable beaver habitat that will reduce the need to forage further from the watercourse and does not create a conflict where flooded (Gaywood, 2015). A unique method indicated by Macdonald et al. (1995) to reduce overall felling rates is where beavers have ring-barked a large tree, is fell the ring-barked tree immediately, as the foliage from it may potentially feed a family of beavers for several months.

5.1.2 Grey squirrel (Family Sciuridae)

5.1.2.1 Background and ecosystem services

Grey squirrel is a non-native invasive species that has an almost complete distribution in England and appears to displace native red squirrels which is well reported (Matthews et al., 2018). Grey squirrels have a strong association with broadleaved woodland (Matthews et al., 2018) and, as with smaller rodents, grey squirrels are 'scatter-hoarders' (i.e. their seed storage locations are dispersed and often with a single food item). Thus, they can be an important seed disperser for English oak (Dyderski et al., 2020), beech (Packham et al., 2012) and hazel. In the latter can help facilitate hazel colonisation onto open ground (Laborde and Thompson, 2009). Grey squirrels may compete for seed resources with native birds and hazel dormice, as well as their well-described impact on red squirrel distribution in the UK (Lever, 2010; The Welsh Government, 2018; White et al., 2011). A review by Broughton (2020) concluded that they "*are unlikely to be significant predators of or competitors with nesting birds in their present or projected range in Europe*".

Mountford and Peterken (2003) suggest that damage by grey squirrels may in the longer term produce what they call 'squirrel pollards' which may mimic somewhat the traditional pollarded woodlands, although this is uncertain. In woodland managed for biodiversity, increased standing and lying deadwood as a result of squirrel damage could have biodiversity benefits (Isted, 2014; Mayle, 2004). However, there is some concern that grey squirrel activity may hinder development of old growth characteristics (Woodland Trust and Hotchkiss, 2020). It has been suggested that grey squirrel damage reduces the ability to use woodland for carbon sequestration (The Welsh Government, 2018). The UK Government (2021) also indicate grey squirrel as a threat to resilience of woodland to climate change and other threats.

5.1.2.2 Damage and impacts

Damage of trees by grey squirrels in the form of bark stripping has been recognised as a problem in production forestry dating back many decades (e.g. Middleton, 1931). In a survey of 441 UK forest managers from 2021, those who managed oaks identified bark damage (including that of grey squirrels) as the biggest threat compared to aspects such as disease, drought and environmental change, and concerns were expressed both about bark stripping and impacts on regeneration (Bates et al., 2024). A survey (n = 777) by the Royal Forestry Society (2021a) indicated many practitioners considered grey squirrels a bigger threat to broadleaf woodlands than pathogens or deer. Although bark stripping is by far the biggest concern relating to grey squirrels, Middleton (1931) noted that grey squirrels can also damage young spruce and larch via biting off buds and shoots, but the extent of such damage is unclear.

Based on data from 2000, DEFRA and Forestry Commission estimated the cost of grey squirrel damage to be up to £10M per year (about £13M today) (Forestry Commission and DEFRA, 2006; Mayle and Broome, 2013) although such estimates are challenging and do not account for other negative impacts such as that on biodiversity. Huxley (2003) refer to quantification of damage to the National Forest Estates published in 2002 of £2M (about £3.5M today) for sycamore, beech and oak, and about £220K (about £400K today) for damage to conifers. White et al. (2011) report figures from Forest Enterprise indicating grey squirrel control costs for their rangers in the mid-1990s were about £330K (about £670K today). A report from 2021 by the Royal Forestry Society estimates the cost of grey squirrel damage in England and Wales at between £1.1M and £45.0M pa across a range of scenarios from low to high severity and the cost of control across the same scenarios from low to high cost as between £1M and £45M (Richardson et al., 2021). Further potential restocking cost for damaged woodlands of up to £19M (Richardson et al., 2021). The report concluded that the most probable cost from those scenarios is a total of £37m a year in lost timber value, reduced carbon capture, damage mitigation and trees to replace those that have died because of grey squirrel bark stripping. It assumes that 15% of the broadleaf area and 5% of the conifer area are damaged or killed by grey squirrels. Growth deformation and staining due to fungal infection secondary to squirrel damage can reduce timber value, and damage to the crown can reduce timber yield (Mayle and Broome, 2013). Mayle and Broome (2013) do note that for woodland intended for biomass production, where quality is less important than quantity, squirrel damage may be less of an economic factor compared to woodland for high quality sawn timber. The cost effectiveness of management for grey squirrels on a local basis relative to the end market value of the timber is challenging to estimate because damage can occur after estimates are made of the final tree crop (The Welsh Government, 2018).

As well as the economic costs, in a social study of practitioners involved in squirrel control Crowley et al. (2018) found that damage by squirrels can have a high emotional cost. As it often occurs on mature trees that people may feel more of a connection to, and severe damage can occur in a short period following years of growth (Huxley, 2003). Damage occurs in both commercial and amenity woodlands (Lever, 2010). Damage may also alter landscape character, for example Rayden and Savill (2004) suggest that the traditional beech wood landscape of the Chilterns is likely to be compromised long-term by squirrel damage and economic decisions not to restock this species as a result.

Bark damage by grey squirrels has been observed in more than thirty species in the UK, but notably sycamore, beech, oak (though in some areas oak seems less susceptible: Rayden and Savill, 2004) and sweet chestnut. It often affects more mature trees of 10-40 (and up to 60) years old, but smaller saplings can be damaged, however conifer damage is uncommon (Gill, 1992b; Mayle and Broome, 2013). A national scale survey based on NFI survey data concurred that sycamore and beech were particularly susceptible, and that damage could affect trees particularly from their second decade onwards (Peden et al., 2000). It affects both planted and naturally regenerated trees (Huxley, 2003). Red squirrel bark damage in the UK has been observed in Scots pine, lodgepole pine, Norway spruce and European larch (Gill, 1992b). Kerr (1995a) noted that ash does not appear to be susceptible to squirrel damage, but Gill (1992b) contradicts this. Bark stripping activity by grey squirrels can result in ring barking of a tree and in a study of an oak plantation in England, affected up to 17% of annually (Mayle et al., 2009). In addition, the damage can impact on the leader and nearly a third of trees had lost their leader or it was vulnerable to dieback due to damage (Mayle et al., 2009). Loss of a leader can change growth form of the tree (Huxley, 2003). In many cases, bark stripping can lead to tree mortality (Mountford, 2006). Damage impacts vary widely but there are many observations in which a majority of trees in a woodland have evidence of damage (reviewed in Mountford, 2006; Rayden and Savill, 2004).

Bark stripping by grey squirrels on thinner stems can girdle the tree and cause stem breakages (Gill, 1992b). There is some evidence that damage is more likely on dominant canopy trees and trees on edges of stands, as opposed to suppressed trees. This may be related to those trees being more actively growing, and trees with thicker phloems are more vulnerable (Gill, 1992b; Kenward and Parish, 1986). In a study of naturally regenerating oak in England, Mayle et al. (2009) found that dominant trees, and those greater than 7.5 cm DBH were at greater risk, and found damage tended to occur higher up (concentrated about 4 m high). Mayle (2004) suggests bark stripping can occur in younger trees if they can support the weight of a squirrel. Based on damage survey data Wait (2008) suggests that onset of damage is related more to size and suitability than age. This is similar to patterns seen in broadleaved woodland on the England-Wales border although in that study there was also a substantial amount of severe damage below 2 m (Mountford, 2006). A bias towards damage in the upper trunk branches of some tree species may be due to those species having bark which thickens or roughens with age lower down the tree that makes bark stripping more difficult (Gill, 1992b). This has also been noted in oak (Huxley, 2003), although Mayle and Broome (2013) note damage lower down can occur in older trees. There is a bias towards damage in thin-barked species (Mayle and Broome, 2013). However, sycamore which has difficult-to-remove bark is still highly susceptible, so ease of bark removal alone is not a principal predictor of damage risk (Gill, 1992b).

The question as to why grey squirrels strip bark has been asked for some time and numerous hypotheses have been posed (Gill, 1992b; Kenward, 1981; Kenward and Parish, 1986). This question has a practical motivation as this knowledge may help in mitigation strategies (Kenward, 1981). It does not appear to be an issue in their native Nearctic range (Huxley, 2003; Nichols et al., 2016). Hypotheses around food shortages, water access, trace nutrient content, sugar content and territorial or agonistic behaviour have been discussed and in many cases there are reasons to doubt these explanations (Kenward and Parish, 1986). Squirrels do appear to select areas/trees with high phloem volume, and this increases the sugar intake per unit area stripped (despite thicker phloems tending to have lower sugar concentration). Kenward and Parish (1986) do cite sources that bark stripping occurs even in captive squirrels that have ad libitum food availability. There is some evidence that damage is highest when there are more juvenile squirrels in a population (which often follows a good mast year) (Kenward and Parish, 1986; Mayle and Broome, 2013; Nichols et al., 2016). More recently, it has been hypothesised that bark stripping is driven by calcium deficiency in grey squirrels. Tree phloem tissue contains calcium oxalate and the hypothesised peak in demand for calcium (via juvenile bone growth and female lactation) would appear to overlap somewhat with the summer bark stripping period (Nichols et al., 2016). However, small-scale laboratory trials have suggested squirrels may not be able to utilise this form of calcium, unlike rabbits, horses and sheep, and further research has been commissioned and is ongoing (Centre for Forest Production, 2025; Nichols et al., 2018)

As summarised by Crowley et al. (2018) squirrel damage creates uncertain economic returns on plantations and may impact on decision-making for restocking. Rayden and Savill (2004) suggest restocking of felled beech with the same species would be uneconomical in the Chilterns. In a survey of 695 woodland managers, grey squirrel damage was frequently mentioned as undermining both the economic and environmental value of new woodlands (Royal Forestry Society, 2021b).

Squirrel damage can increase risk of tree disease and fungal infection and grey squirrel damage has been linked to increased infection by *Phytophthora ramorum* in larch in Wales and Northern England (Mayle and Broome, 2013) although links to impacts in oak are less evident (The Welsh Government, 2018). Fungal infection secondary to squirrel damage can lead both to mortality and downgrading of timber products (Huxley, 2003; White et al., 2011; White and Harris, 2002). Callousing over wounds will frequently not cover the whole area leaving exposed wood (Huxley, 2003). Callousing can also lead to fluting which can reduce value of timber (Huxley, 2003). Structural strength of affected timber can also be reduced (Mayle, 2004) and if poorer quality timber were to be used for firewood instead of as structural timber, this could have consequences for overall carbon budget. Squirrels can act as seed dispersers (discussed above) but also predate on seeds and dig up germinating seeds such as oak (Harmer, 1995). Selective seed predation could impact on the tree species composition of native woodlands (Woodland Trust, 2013a).

5.1.2.3 Identification of damage

Grey squirrel damage is primarily in the form of bark stripping and can occur both in the upper portion of the trunk or high branches, or the base of the trunk and the root buttresses (Gill, 1992b). Red squirrels can strip bark occasionally but this tends to be in spirals around the upper trunk (Gill, 1992b). Such bark damage typically occurs in late spring or early summer, and less so in later summer (Gill, 1992b) although it can extend into September (Mayle, 2004). This can help distinguish lower trunk squirrel bark damage from basal bark stripping by rabbits which tends to occur over winter (Springthorpe and Myhill, 1994). The wounds caused by grey squirrels can reach more than 1 m^2 in area (Gill, 1992b). A characteristic of squirrel bark stripping is that the outer bark is typically not consumed so may be left in coils on the tree or chips on the ground (Springthorpe and Myhill, 1994; The National Forest, 2020). Squirrel teeth marks are 1.5 mm wide in pairs and typically run parallel to the stem of a tree (Forestry Commission, 2023).

Huxley (2003) also reports the phenomenon of squirrel ‘trials’, i.e. small hand-sized patches of bark stripping that have been observed following thinning as precursors to more extensive damage (and possibly represent squirrel assessment of phloem volume). Forestry Commission (2023) refer to these as “tester patches” and they can be as small as a coin. Grey squirrel damage is potentially confusable with that of edible dormice which also strip bark of similar tree species in early summer (Gill, 1992b). Kenward and Parish (1986) used a six-point scale to categorise extent of damage which was based on both size of stripped patches and proportion of trees impacted, Mountford (2006) a four-point scale and Rayden and Savill (2004) a five-point scale. The National Forest (2020) have a 5-points cale for damage with photo examples provided.

Grey squirrel predation of hazelnuts can be distinguished by a zig-zag pattern breakage of the nutshell and a hard twist to open the shell, compared to a neat round hole with tooth marks made by mice or voles (Laborde and Thompson, 2009).

5.1.2.4 Protection against damage

Tree protection methods identified in the literature for grey squirrels are shown in Figure 9, with details and evidence base synthesised below.

MONITORING BEFORE INTERVENTION

Mayle and Broome (2013) describe a monitoring regime that can help to predict if lethal control might be necessary in a local area. This involves winter trapping and drey surveys to predict the availability of food and the breeding status of the local population. Where there is evidence of a squirrel population with high food availability and early breeding, this predicts high damage risk and need for lethal control. A similar approach is outlined in Gill (2019) which is based on several factors such as autumn seed availability, winter breeding, and existing damage levels. Such an approach has been called ‘index trapping’ (Huxley, 2003). Huxley (2003) report a study in which cost of lethal control based on monitoring and targeting a minimal intervention policy reduced cost by 60% over 5 years. This used information such as management history (e.g. if thinning has occurred recently, presence/quantity of ‘trial’ stripping patches and masting quantity).

Springthorpe and Myhill (1994) also identify risk factors for deciding on control as high squirrel numbers, which species may have been damaged in recent years (which might predict damage in the next) and good mast years that tend to increase numbers and subsequent bark damage. Forestry Commission (2023, 2022), Woodland Trust (Hotchkiss et al., 2022) and The National Forest (2020) provide advice on grey squirrel impact assessment. Gill (2019) provide a decision tree in terms of overall control strategy that starts with categorisation of damage risk and helps decide whether monitoring or intervention (and then which type of intervention) should be implemented. In terms of monitoring squirrel numbers themselves, Gurnell et al. (2009) provide data a practical guide on expected carrying capacities for different habitat types, and what survey methods (visual, hair-tubes, drey counts, feeding signs and whole maize bait) are appropriate in different situations. They note that the best survey tool may depend on the incidence of signs of squirrels and which species (grey, red or both) are present. Beatham et al. (2023) demonstrate that an index of grey squirrel camera-trap images was highly correlated to the density of squirrels trapped and removed across ten woodlands in England and Wales, suggesting a useful potential monitoring method to inform future risk assessment and decision making.

REDUCE TREE RESOURCE DEMAND

It has been stated that grey squirrels suffer virtually no natural predation in the UK (Gill, 1992b; Middleton, 1931). However, reintroduction and range expansion of pine marten offer a potential form of biological control to reduce grey squirrel densities and damage. In Ireland, a range increase in pine marten *Martes martes* (via increased habitat availability and legal protection) has correlated negatively with grey squirrel populations (and positively with red squirrels) and other causes of grey squirrel decline are not evident (Sheehy and Lawton, 2014). Pine martens do seem to be more likely to prey upon grey squirrel than red, even where the latter are numerous, but population changes cannot be accounted for by diet alone (Sheehy and Lawton, 2014). Negative associations between pine marten and grey squirrel could operate through an indirect mechanisms including change in foraging/feeding behaviour influencing body condition and reproductive success (Sheehy and Lawton, 2014). This might be important given associations between juvenile densities and bark stripping discussed above. In Scotland, as part of grey squirrel control to protect red squirrel populations, artificial pine marten dens are being installed within forestry as part of a plan for encouraging negative impacts on grey squirrels (Forestry and Land Scotland, 2024, 2022a). There is currently a long-term strategic recovery plan for pine martens in Britain which may reduce grey squirrel numbers longer term, but it remains to be seen if this would have the effect of reducing or eliminating damage to trees and how this may vary geographically (MacPherson and Wright, 2021). Under present knowledge this approach cannot be relied on to be a solution (The Welsh Government, 2018). Goshawks are also predators of grey squirrels, and their breeding season may coincide with the time when young squirrels are leaving dreys (Field, 2023; Henderson and Conway, 2017). Field (2023) suggests that management approaches that benefit existing goshawks via reduction in disturbance or that encourage their colonisation of woodland (for example providing stands of conifers away from public footpaths that is attractive nesting habitat) may assist in managing squirrel populations. There is some suggestion that gene drive could be a potential long-term solution to grey squirrels in great Britain, although this is still in early exploratory stages (Hartley, 2024). Gene drive is a form of genetic engineering to 'drive' a particular genetic trait through a population to the point that 100% of a population will eventually carry a specific gene (Kaiser, 2021). The approach being investigated is Directed Inheritance Gender Bias (DIGB), which would skew the sex ratio of offspring the grey squirrel population leading to population crash (Faber et al., 2021; Kaiser, 2021).

Recently, contraception has been investigated as a potential tree protection method against grey squirrels (Baker, 2010). A questionnaire study by Barr et al. (2002) suggested this approach would be welcomed by practitioners involved in grey squirrel control and relative to lethal methods (shooting, trapping and poisoning) was considered overall as a more 'humane' approach. Among the UK general public (n = c. 4,000), use of contraceptives was considered generally more acceptable as a control method compared to several lethal control approaches (Dunn et al., 2018). The only feasible delivery system for such contraceptives is via oral baiting (Moore et al., 1997). Beatham et al. (2023) examined the effectiveness of different baits at squirrel feeders towards maximising uptake of contraceptives. They found that hazelnut butter would allow delivery to the majority of squirrels within four days, and that bait uptake was higher in summer compared to winter and where feeder density was higher. An individual-based modelling study by Croft et al. (2021) based on data from Cumbria predicted that at high grey squirrel densities provision of contraceptives would be unlikely to act quickly enough to be able to replace culling as a damage reduction method, but it could work on lower density populations following short-term culling efforts.

Contraceptive drugs that have been proposed or trialled in grey squirrels include immune-contraceptive vaccines (Moore et al., 1997) and cholesterol mimics that block steroid hormone production (Mayle et al., 2013; Yoder et al., 2011). Both of these inhibit fertility of both males and females. The contraceptive vaccine (albeit via injection) has been shown to have some success in other non-squirrel species in the wild, and an oral dose has been shown to be successful in rats in laboratory studies (UK Squirrel Accord, 2021). Palatability of these contraceptives are undergoing trials (UK Squirrel Accord, 2021) Beatham et al. (2024) carried out field trial in six English woodland populations, deploying bait via bespoke squirrel bait hoppers with integrated PIT-tag readers. They were able to deliver multiple doses on most days to most males and females, and this was particularly effective in spring and winter, which correspond to immediately before the second and first peaks in breeding respectively (Beatham et al., 2024). Consideration is also being given to whether contraceptive baits can be used within existing warfarin feeders to reduce wastage and cost, whilst work is also being carried out on hopper designs that use a weighing platform to ensure access only by the target species (UK Squirrel Accord, 2021). Mayle et al. (2007) provide advice on preventing use of hoppers by non -target species such as game birds, which although were intended for poison bait, may be useful for preventing non-target consumption of contraceptive baits too. It should also be noted that where pine martens are present several control measures such as baited hoppers but also drey poking, spring traps and Goodnature traps should be avoided (Gloucestershire Wildlife Trust, n.d.)

Diversionary feeding is not considered an effective control method for grey squirrel damage as it can increase numbers and thus damage, and there is no apparent link between grey squirrel damage to bark and scarcity of food supply (Gill, 1992b). Kerr (1995a) does suggest that sycamore and maple which are favoured by grey squirrel could be used as palatable diversion (or 'decoy') trees but a benefit of this is not tested. This method is already being implemented in some new planting schemes with about 5% of trees on one scheme being planted as 'sacrificial' sycamores (Royal Forestry Society and Forestry Commission, 2022).

Despite various regional and localised attempts (e.g., in Thetford Forest: Mill et al., 2020; various schemes also reviewed in Sheail, 1999), eradication seems unlikely to be a solution to tree damage in the UK (Baker, 2010). National eradication schemes can be unpopular with some members of the public, as happened in Italy when grey squirrel eradication was attempted (Bertolino and Genovesi, 2003), but local and short-term lethal control is the most common method used, and can consist of live trapping (with humane dispatch), spring trapping (i.e. immediately lethal), shooting (Baker, 2010). In the past poisoning, principally with warfarin was common but this is no longer licenced for use. The control approach taken and the placement of traps might be influenced by the extent of public access (Gill, 2019; The Woodland Trust, 2019). Lethal control is also often motivated by efforts to protect red squirrels which are negatively associated with grey squirrel populations (Baker, 2010), as well as to protect trees from bark damage.

Bark damage can occur when squirrel densities are at 4-5 squirrels per hectare (Mayle, 2004). Densities often far exceed this, and it has been suggested that the most economical approach to reducing damage is to concentrate lethal control to sites most vulnerable to damage just prior to and during the May-August period of damage with an aim of reducing densities to < 5 per hectare (Mayle and Broome, 2013). Since increased growth rates of trees following thinning are a risk factor for bark damage (as the phloem volume can increase) then planning for lethal control for this period (or monitoring for early signs of damage such as tester patches) may be critical (Hotchkiss et al., 2022). Short term but intensive control at the optimum time will be more effective than lower intensity continuous control because factors such as density dependent reproduction and dispersal can compensate for such reductions (Huxley, 2003). In some cases, if local eradication is achieved, it can take several years for densities capable of damage to reappear (Huxley, 2003). Coordinated control efforts involving cooperation with neighbours, for example via Squirrel Control Groups, as can be effective for deer culling, may also increase effectiveness by reducing chance of rapid recolonisation following control (Gill, 2019; Huxley, 2003). Recolonisation can happen in just a few weeks (Mayle and Broome, 2013) and potentially could actually increase damage (suggested by Woodland Trust, 2013a). Gill (2019) also suggest targeting control on any dispersal corridors that would allow recolonisation of an area post-culling. It is also suggested that control is not just targeted at areas directly receiving damage, but surrounding areas that may also be used by the same squirrels (Mayle, 2004). Mayle (2004) gives an estimate of control costs of between £7-11 per ha (about £12-19 today) depending on control method and other variables. Another source quotes anecdotal evidence that it can reach £70 per ha ("anecdotal evidence" reported by The Welsh Government, 2018). Squirrel control is a long-term commitment for a commercial timber crop because the period of vulnerability of trees may last several decades (Mayle et al., 2007). This represents a significant financial and time commitment for small-scale woodland owners (Royal Forestry Society, 2024a)

The EU license for the production and sale of warfarin as a grey squirrel bait ended in September 2014, but users who had stocks left could use it until September 2015 . Anecdotal evidence may suggest that squirrel numbers have increased in areas it had previously been used as a control method (Richardson et al., 2021). Not long before its cessation, a survey suggested it was one of the most widely used methods and relatively inexpensive compared to trapping and shooting (Isted, 2014). It was an effective control method (Mayle et al., 2007) but could not be carried out where either red squirrel or pine marten are present due to risk of non-target impacts (Mayle and Broome, 2013; Sheail, 1999). It was typically achieved through warfarin treated grain placed in specially designed hoppers (Rayden and Savill, 2004). However, Huxley (2003) pointed out there was a risk to non-target species both via direct consumption of the poison and via secondary poisoning (including sublethal effects). Huxley (2003) reported results from a trial that suggested poisoning could be effective at reducing grey squirrel number substantially in 4 weeks if hoppers were spread widely and adequately pre-baited. That document provides a sample control programme with details on timing and density of hoppers (typically 1 per hectare).

Live trapping of squirrels has to be accompanied by humane dispatch, for example by a blow to the head (described in Crowley et al., 2018) or shooting (Gill, 2019). Trapping is generally considered to be very labour intensive as traps must be inspected at least once daily (l). However, since banning of Warfarin for grey squirrels (UK Squirrel Accord, 2021) it remains perhaps the best current solution. Where red squirrels are present, it is suggested live traps are visited every four hours and if pine marten are present a restrictor narrowing the entrance to 4.5 cm is advised (Gill, 2019). Labour is probably the highest cost involved in grey squirrel trapping (The Welsh Government, 2018). There are a variety of live trap designs on the market, including single- and multi-capture traps, and also remote trap monitoring systems are available (British Red Squirrel, 2019; Gill, 2019). Where red squirrels or pine martens are present, single-capture traps should be used (Forestry Commission, 2023), and placing traps in trees on platforms may also reduce chance of non-target species capture (Gill, 2019). In terms of welfare of animals within traps, as well as regular visits, protecting against risks of flooding, hyper/hypothermia, and disturbance by domestic animals, people or potential predators is important (Northern Ireland Squirrel Forum (NISF), 2024).

Lethal trapping has undergone technological developments, for example the GoodNature trap which was recently licensed (The Spring Traps Approval (England) Order 2018) is designed to kill squirrels using a gas-powered bolt through the cranium and allows multiple kills because the targeted squirrel drops to the ground and the trap resets (Crowley et al., 2018). However, in a recent survey of woodland managers, this type of trap was ranked lowest in terms of effectiveness, although it is noted as it is relatively new more trialling of best operational use may be required (Royal Forestry Society, 2021a). However, this trap has been modified since first introduction and best practice work has been done but remain unpublished (Rebecca Isted, Forestry Commission, pers. comm.). Nevertheless, more traditional spring traps (often placed in tunnels) are still widely used and effective and must be checked once per day (The Welsh Government, 2018); bird by catch can be reduced by using baffles (Gill, 2019). In some areas, trap loan schemes operate which may reduce cost to individual forest managers (The Welsh Government, 2018). Generally live trapping is suggested as being safer in terms of reducing non-target catch but has been suggested that spring trapping can kill squirrels that are trap shy and may not be caught in live traps (Gill, 2019). For trapping programmes choice of sites for hoppers or traps can be critical to success. Both knowledge of locations frequented by squirrels and pre-baiting of sites are important in this (Gill, 2019; Springthorpe and Myhill, 1994). Mayle et al. (2007) give some site choice suggestions such as under the largest trees with branch tips extending to near the ground, and tree stumps.

Mayle and Broome (2013) and Huxley (2003) suggest that shooting (often combined with drey poking, which involves using a long pole to flush squirrels out of their dreys) alone may not be effective at reducing densities sufficiently during the damage period since this typically has to be carried out during winter or early spring. At this time, there is no leaf cover which leaves sufficient time for dispersal into the area before or during the period of damage. However, shooting could be used as an additional, complementary component of a control programme (Huxley, 2003; Isted, 2014) and Forestry Commission (2023) also suggest it is most effective when combined with trapping. It should be noted, however, that more advanced approaches for shooting are being developed that counter the claim of reduced effectiveness of shooting for grey squirrel control. Braithwaite and Dutton (2023) outline air rifle methods that are different to traditional drey poking approaches and involve use of feeders and bait stations and thermal imaging approaches. They argue that the energy of air rifle pellets at typical distances (25-40 yards, facilitated by use of hides), and precision of modern air rifle equipment are sufficient to enable humane killing. However, it should be noted that this study does not evidence the impact on tree damage. They used camera traps to help narrow down peak usage times at feeders and thus increase efficiency of effort and field trials demonstrated a reduction in grey squirrel numbers over several years (Braithwaite and Dutton, 2023). These approaches are also described by Gill (2019).

A consultation survey (n = 136) by Isted (2014) suggested combination lethal control methods (such as trapping followed later in the year by shooting) were considered most effective. A survey (n = 777) by the Royal Forestry Society (2021a) indicated many practitioners considered no control methods very effective and indicated the need for more training. Shooting at bait stations and use of Fenn traps were considered the most effective methods, with single-catch live traps in third place.

MODIFY ACCESS TO TREE RESOURCES BUT NOT AVAILABILITY

Generally, methods to physically exclude squirrels from trees are ineffective since they can easily climb and commute across the canopy layer. As with lagomorphs and voles, the chemical repellent Aaaprotect has potential for protection for bark stripping according to Hodge and Pepper (1998), but for ground species the area coverage can be relatively small (for example up to 30 cm for voles). Thus, it is unclear how practical this would be for grey squirrels that typically strip bark higher and in pole stage or mature trees.

MODIFY TREE RESOURCE AVAILABILITY AND RESILIENCE TO DAMAGE

Landscape placement of woodlands could have potential for prevention of grey squirrel damage, for example in new farm woodlands. Fitzgibbon (1993) found in East Anglia that across 68 deciduous woods (in a landscape with relatively low woodland cover) probability of grey squirrel occupation of woodlands was reduced in smaller woods, isolated woods (particularly those further from larger woods), containing less oak, beech or hazel, and that had less hedgerow habitat around them (the latter of which may facilitate movement of squirrels between woods). They suggest that new deciduous woodland that is placed at least 0.5-1km away from the nearest large (5 ha or larger) woodland and where there are fewer hedges might reduce risk of squirrel damage risk. In addition, the suggest avoiding placing new woods near to woods with hazel or beech. Huxley (2003) reports a large beech wood planted under these criteria in the Chilterns that had not shown signs of squirrels or damage. In the context of protecting red squirrels in Cumbria, Stevenson et al. (2013) applied least cost analysis to look at habitat networks and dispersal corridors for grey squirrels, as a way of identifying areas that could target management to prevent spread over the landscape. It is possible such models could also be used in decision making for placement of new broadleaf woods within landscapes to minimise risk of colonisation by grey squirrels or indicate where control might reduce risk of colonisation. It should be noted that radio-tracking of grey squirrels by Stevenson et al. (2013) did indicate that squirrels could move across gaps between woodland patches of scale suggested by Fitzgibbon (1993) had suggested would create effective isolation.

Mountford (2006) suggest that tree species choice and unpalatable planting could be a solution to squirrel damage, for example more of a focus on growing ash and lime which are less susceptible. It should be noted that currently the movement of ash planting material is prohibited so ash can only be planted if there is a source of seed on site (Forest Research, n.d.). Rayden and Savill (2004) also suggest ash or cherry as suitable choices for unpalatable planting. In a consultation study (n = 136) 44% of respondents has altered tree species planted in response to grey squirrel impacts. In some areas, specific species are deliberately being avoided due to their attraction for grey squirrels. For example, oak is being avoided in one upland planting scheme, although the motivation in that case is protection of red squirrel populations (Royal Forestry Society and Forestry Commission, 2022)

Another protection solution proposed by Mountford (2006) was growing vulnerable species such as beech as an under-storey or in a small group or selection (continuous cover) system with limited thinning until the trees are over 35 cm dbh. Suitable nursing species proposed include shade tolerant western hemlock or western red cedar (Huxley, 2003). The argument for this relates to observations that damage tends to effect faster growing individuals (Gill, 1992b; Kenward and Parish, 1986) and slowing growth rate may reduce damage risk. However, these approaches may reduce the profitability if used in commercial plantations (Huxley, 2003).

5.1.3 Small mammals (Families Cricetidae, Muridae and Gliridae)

5.1.3.1 Background and ecosystem services

There are a few small rodent species relevant to tree and treescape damage in a UK context: field vole, bank vole, wood mouse, yellow-necked mouse and edible dormouse. Bank and field voles and wood mice have fairly widespread ranges in England, while yellow-necked mouse is more restricted to the south and midlands (Matthews et al., 2018) The introduced edible dormouse is restricted to a small area of the Chilterns. A key ecosystem service related to small mammals is seed dispersal. Scatter-hoarding wood mouse and yellow-necked mouse can aid seed dispersal of English oak and hazel over short distances (Dyderski et al., 2020; Laborde and Thompson, 2009). Wood mice and voles can disperse seeds such as beech, cherry and hornbeam via caching, which can aid in the process of colonisation of new woodland (Forestry Commission, 2021). As with lagomorphs, small rodents are important prey items for many species of carnivorous mammal and birds of prey.

5.1.3.2 Damage and impacts

The impacts of voles and mice on young plants in nurseries and plantations has been noted for at least a century (Middleton, 1931). Gill (1992b) suggest that due to differences in habitat preferences, the two main species of vole may have different effects on trees. More open habitat-associated field voles causing damage in newly planted land or large areas of restocked forest (presumably when field vole populations have grown during a fallow period), while bank voles more likely to impact natural regeneration within woodlands or small replanted areas. This pattern is supported by a study of young commercial beech plantations in the Czech Republic surrounded mostly by spruce monocultures. Although bank voles were more common, vole damage to the plantation trees was almost exclusively by field voles (Suchomel et al., 2016). For both species, bark stripping is the primary cause of damage to trees, particularly to young trees and in winter, and this can also affect roots under ground level (Gill, 1992b). However, seedling buds and stems can also be browsed (Gill, 1992b). While bank voles are known to climb up to 10m to remove bark from a main stem, this is apparently rare in Britain (Gill, 1992b). Climbing of saplings by bank voles to feed on terminal shoots has also been reported from Russia (reported in Pigott, 1985). It has also been noted that bank voles can remove buds, particularly of newly planted pine trees (Forestry Commission, 2023). Rogers Brambell (1958) report damage at 1-2 m above ground to be confirmed by trapping as from bank voles climbing. Gnawing by field voles at the base of saplings can actually fell the tree, and this has been reported in an oak sapling of 2.7 cm diameter (Davies and Pepper, 1993).

While there can be a lot of interannual variation in damage, probably linked to vole population cycles, and damage can be patchy and localised, in extreme cases new commercial plantations have suffered 100% losses (Gill, 1992b). In the Czech Republic vole damage frequently led to sapling mortality of a range of species, and more frequently led to mortality (particularly via girdling) than deer fraying or hare browsing (Krojerova-Prokesova et al., 2018). Mortality due to vole bark gnawing and browsing was also extensive in some other broadleaved species (Including lime, oak and beech) in England (Pigott, 1985). Andrews et al. (2000) reported 73% loss of seedlings regenerating on open moorland in one study in Scotland, due to voles eating the main shoots. In Yorkshire, in some areas of an ex-agricultural upland site being modified to native woodland, about one third of planted trees were lost due to vole damage (Forest Plastics Working Group, 2024a). Apart from mortality, if vole damage girdles the leader, this can lead to multiple stems or crooked based trees, although in conifers such as Sitka spruce a fresh leading shoot may be produced from epicormic growth beneath the damage (Rogers Brambell, 1958).

Vole cycles in Europe tend to be stronger at higher latitudes, particularly > 60°N but most economic damage occurs further south (40-60°N) where vole cycles conflict with commercial plantations (Jacob and Tkadlec, 2010). Field and bank voles tend to have cycles of 3-4 years and 3-5 years respectively although there has been evidence of weakening of the cyclical pattern for field voles, perhaps due to climate change (Jacob and Tkadlec, 2010). The relationship between damage and population peaks may not just be related directly to numbers, but also nutrient deficiencies in individual voles when numbers and thus competition for food are high (Gill, 1992b). In a study in the Czech Republic, a key factor in presence and abundance of *Microtus* voles (including field vole) in and around forestry plantations was the occurrence of grasses (Krojerova-Prokesova et al., 2018). Dense grass swards provide ideal cover for voles (Hotchkiss et al., 2022). Ground vegetation biomass was a strong predictor of vole numbers in new farm woodland in Hampshire, (Trout et al., 2004). Vole and mice damage can be a problem where forestry is planted on highly productive ex-arable land where a 'luxuriant' ground layer can mean very high numbers which in turn impact newly planted trees (Trout et al., 2004).

Data on tree preferences are reviewed by Gill (1992b). While some species may be more or less susceptible, a large number of species can be fed on by voles. There is some evidence that oak is avoided by voles (Krojerova-Prokesova et al., 2018). In an experimental study by Pigott (1985) a large amount of bank vole damage occurred on planted small-leaved lime saplings, but not on naturally regenerating birch. Pigott (1985) quantified the rate of damage of captive bank voles (albeit they were also fed additional food) and found they would gnaw the bark of and remove the bud of 2-6 small-leaved lime seedlings per 24h. Although generally younger trees are more vulnerable (or preferred) to vole damage, and Jacob and Tkadlec (2010) indicate that at about 5-8 years old vulnerability is reduced (which may be attributed to a reduction in nutritional value of bark with age). While Trout and Brunt (2014) suggest trees are most vulnerable in the first three years, trees up to 14 years old have been damaged by field voles in the UK (Gill, 1992b). It has been reported that voles can have a preference for male plants in dioecious willows which can skew the sex ratio in a willow population (reported in Gill, 1992c).

Where direct sowing is used, small rodents can be the primary cause of losses (Armour, 1963). Voles and mice can be responsible for seed predation in nurseries or direct sowing scenarios (Rogers Brambell, 1958; Trout et al., 2004). This can be a factor in limiting success of planting schemes (Parratt and Jinks, 2013). In the context of natural regeneration, the reduction in seeds does not appear to impact germination in beech (Packham et al., 2012) or hazel (Laborde and Thompson, 2009). However, in situations of densely hand sown seeds predation rates can be very high (Packham et al., 2012). Similarly, hand sown small-leaved lime seeds suffered almost 95%

predation by rodents (with wood mice indicated as the most likely species) (Pigott, 1985). Trout et al. (2004) also report a 100% failure rate of hand sown ash within windblown beech woodland due to small mammal seed predation. Pigott (1985) report stomach content analyses suggesting that seeds are a more common part of the diet of wood mice relative to bank voles, the latter of which feed more on foliage, although will feed on small tree seeds (Andrews et al., 2000). Both bank voles and yellow-necked mice will consume fir seeds (reported in Senn and Suter, 2003). Juniper seeds are highly palatable to wood mice, to the extent that direct sowing without protection is unlikely to be successful, but also possibly to bank and field voles (Packham et al., 2012). Parratt and Jinks (2013) carried out an experimental study of seed preferences of small mammals in the UK and found distinct preferences that related to the size of seeds and the level of protection offered by that seed species. Oak, beech, sycamore and hazel were preferred and often completely removed within a few days (within one day in the case of oak). Wild cherry, hawthorn and ash were least preferred. In most cases dry weight of seed was the best predictor of its degree of preference (heavier seeds being mor preferred).

Edible (or fat) dormice are still restricted to an area in the Chiltern Hills where they were first introduced in 1902 (Baker, 2010; Matthews et al., 2018). They have been subject to at least one failed eradication attempt (Mill et al., 2020). Like grey squirrels, edible dormice can cause stem breakage in tree crowns due to bark damage (including full girdling of the main stem) which can cause forking of trees or suppression of a tree by undamaged or less damaged neighbours (Gill, 1992c). Bark removal by edible dormice tends to occur in early summer, and typically from them main stem in the crown of a pole or mature stage tree, and the damaged patches can have a characteristic rectangular shape (Gill, 1992b). There do seem to be tree species preferences, with larch, Scots pine, Norway spruce, beech and birch affected, but apparently not sycamore or oak (Gill, 1992b) and also damage to orchard trees (White and Harris, 2002). Edible dormice will also consume beech nuts (Packham et al., 2012).

5.5.2.3 Identification of damage

Harmer (1995) suggest that vole damage may sometimes be mistaken for rabbit damage. There are distinctions between field voles and bank voles that can aid identification. Observations from the Czech Republic suggest that the *Microtus* genus (containing field voles) in general gnaw deep into stems, leaving the wood inside visible, while bank vole damage can be more superficial, leaving the darker phloem and cambium layers visible (Krojerova-Prokesova et al., 2018; Suchomel et al., 2016). In addition, generally damage above 20 cm can be attributed to bank vole and damage below 20 cm to field vole (and below 10 cm has also been suggested: Trout and Brunt, 2014). However, if snow causes saplings to bend, even field vole damage can occur high on stems (Krojerova-Prokesova et al., 2018; Suchomel et al., 2016). Davies and Pepper (1993) report that field voles rarely climb. Forestry Commission (2023) and Davies and Pepper (1993) state that field vole damage goes up to the height of the surrounding vegetation, rather than assigning a specific height to it. Pigott (1985) states that damage by bank and field voles is very similar and can probably not be distinguished, although this contrasts to most sources that make a distinction.

In the UK damage has been distinguished as 'basal bark gnawing' (by field voles) and 'high barking' by bank voles and both generally occur in winter (Springthorpe and Myhill, 1994). However, Forestry Commission (2023) and Hodge and Pepper (1998) note field vole damage can occur year round (also noted in Davies and Pepper, 1993) and bank vole damage in spring. Springthorpe & Myhill (1994) describe that basal bark gnawing by field voles often occurs just above soil level and often below the grass mat (distinguishing it from rabbit gnawing which is above the grass mat). In addition, they also note that the gnawing is not typically as concentrated as that of rabbits and hares, with areas of bark often remaining between gnawing marks. Forestry Commission (2023) suggest that bark is removed in strips 5-10 mm wide by both species (albeit both can strip around the entire stem). The high barking of bank voles is described as occurring particularly during snow cover and at typical heights of 60 cm to 2 m (Springthorpe and Myhill, 1994). It occurs often at the axil between the main stem and branches of a sapling (probably as this location is easier for the vole to sit) but is less common within commercial plantations (Springthorpe and Myhill, 1994). Pigott (1985) reported the width of individual teeth marks from mice and voles to be 0.8-0.9 mm, Springthorpe and Myhill (1994) indicate width of vole teeth marks as < 1.3 mm, and Forestry Commission (2023) indicate incisor marks are 1 mm wide in pairs. Where teeth marks are quoted as being about 2mm wide (e.g. Forest Research, 2024b), this likely then refers to the width across both incisors.

Bark removal by edible dormice tends to occur in early summer, and typically from the main stem in the crown of a pole or mature stage tree (Gill, 1992b). The damaged patches can have a characteristic rectangular shape (Gill, 1992b) although it can look similar to damage by grey squirrels and there is overlap in when such damage occurs between the two species (see also: Springthorpe and Myhill, 1994).

5.1.3.4 Protection against damage

Tree protection methods identified in the literature for small mammals are shown in Figure 10, with details and evidence base synthesised below.

MONITORING BEFORE INTERVENTION

Jacob and Tkadlec (2010) suggest that development of <u>predictive models</u> of vole abundance could help in reducing damage in this way or targeting other protection methods most efficiently.

REDUCE TREE RESOURCE DEMAND

The Forest Plastics Working Group (2022) indicate that <u>provision of raptor perches or owl/kestrel nest boxes</u> can reduce incidence of vole damage although do not provide evidence for this. Similarly, it has been suggested that <u>leaving any broadleaves standing during commercial conifer harvesting</u> may help reduce vole numbers in later restocks by providing habitat for raptors and owls, but again this is not tested (Royal Forestry Society and Forestry Commission, 2022). In Spain, abundance of common vole (*Microtus arvalis*) has been decreased by providing nest boxes for common kestrel (*Falco tinnunculus*) and barn owl (*Tyto alba*) in agricultural landscapes and fruit orchards (Paz Luna et al., 2020) but in an English context, this vole is restricted to Guernsey and there is no similar evidence for our more frequent tree pests.

Generally, for voles and mice at such high densities, lethal control is not typically a viable option (Davies and Pepper, 1993; Rogers Brambell, 1958; Trout et al., 2004). In Europe, a main control method for rodents, including voles, has been the use of poisons, although there are several factors including impact on non-target species and legal restrictions to this (Jacob and Tkadlec, 2010). Warfarin for use on voles is not approved in the UK (Davies and Pepper, 1993). For edible dormice, which presents a more localised problem, a trapping method that has been developed to control numbers consists of the use of nest tubes (Morris, 2004). These can be made from cheap flat- sided tree tubes/shelters and plywood, and suspended about 2.5-3 m above ground, resembling a hollow branch. In a trial in the UK, nest tubes were occupied within 3 months by edible dormice and 12/20 (60%) were occupied (Morris, 2004). As such, this has been suggested as a means to control them, although this has not been tested (Morris, 2004).

MODIFY ACCESS TO TREE RESOURCES BUT NOT AVAILABILITY

For juniper seed plants, Thomas et al. (2007) reported evidence that 15 cm high mesh tree guards have been successful against mice predation. Spiral vole guards are now widely used and recommended by Forestry Commission in all cases where tree tubes/shelters or mesh guards are not being used (Forestry Commission, 2020a). These are typically 25-30 cm high and need to be pushed into the ground (Trout et al., 2004), by about 5mm according to Davies and Pepper (1993). In addition, it needs to be ensured that any surrounding vegetation leaning towards the planted sapling, or rocks, stumps etc. don't provide opportunities to climb above the height of the vole guard. Davies and Pepper (1993) do suggest that voles can gain access between the spirals of such guards. They suggest that guards with ventilation holes should also be avoided as they create opportunities for voles to get through. The narrower diameter of vole guards (typically about 5 cm) is unlikely to leave sufficient space for voles or mice to nest inside them (Davies and Pepper, 1993). The Forest Plastics Working Group (2022) indicate that while vole guards are cheaper to collect in (when trees have matured) than tree tubes due to lower labour requirement, PVC versions are difficult to recycle. In some cases of mixed species plantations vole guards have only been applied to the more vulnerable thinner barked species such as birch (Royal Forestry Society and Forestry Commission, 2022). Hunt (2024) describes trials of several different vole guards made from more sustainable materials including those made from poly-lactic acid, sugarcane/corn/starch, cotton and wood. They report mixed success with aspects such as brittleness and flimsiness being important factors reducing apparent effectiveness. They also report that unit cost for non-plastic vole guards are typically 2-3 times that of traditional plastic versions.

Small split plastic tree tubes may also be effective against voles (Hodge and Pepper, 1998). It appears voles do not gain access via the overlapping split as they may do with ventilation holes in other guard types (Davies and Pepper, 1993). These designs open up as the tree matures and then can be easily collected, but as with spiral guards, must be buried into the ground by at least 5 mm (Hodge and Pepper, 1998). Larger tree tubes, if not pushed into the ground, can actually encourage small rodents by providing warm shelter and nest sites, and where soils crack in the summer small mammals may be able to travel under any tree tubes or vole guards not pushed sufficiently into the ground (Kerr, 1995b; Trout et al., 2004). Where evidence of voles and mice nesting in tree tubes is found, it can be accompanied by evidence of tree damage within the tube as well (Forest Plastics Working Group, 2024b). In addition, Davies and Pepper (1993) note that any windblow that tilts a tree with a tree guard on, or soil erosion that can occur due to rain running down the tree tube, can leave a gap for voles to enter (voles can enter a gap of 5 mm: Potter, 1991). They suggest that a spiral vole guard can be placed inside the tree tube to achieve vole protection on top of other benefits of tree tubes. This may also provide a double layer of protection since tree tubes can be gnawed through by voles (Potter, 1991). A well attached stake driven firmly into the ground may also reduce the chance of a tree tube tilting due to wind and allowing vole entry (Potter, 1991). It has been suggested that 1.2 m high tree tubes offer protection from edible dormice (Royal Forestry Society and Forestry Commission, 2022).

Cut branches of gorse packed around the base of young trees have been tried but its effectiveness is not reported (Armstrong and Robertson, 2013). It should be noted that in an experimental study in Germany, while the retention of deadwood in the form of lying tree crowns reduced deer browsing probability on silver fir saplings, rodent browsing (identified as most likely bank vole and yellow-necked mouse) actually increased with more lying deadwood (Hagge et al., 2019).

Since small mammal predation on seeds can be a big problem, a variety of different seed coatings have been trialled over many decades (Thompson, 1953). The chemical repellent Aaprotect can be used (via painting or spraying) against voles and requires reapplication each six months (Davies and Pepper, 1993). It needs to be applied on the base of the stem of a tree up to a height of 30 cm for field vole protection (Hodge and Pepper, 1998). Birkedal (2010) tested the effectiveness of using mink excrement as a repellent for acorns and found in laboratory trials reduced bank vole consumption of beech and oak seeds with no or only minor reductions in germination rate.

Ground layer vegetation removal from around trees (via mowing, grazing or herbicide applications are widely used tree protection methods against voles since they reduce the suitability of habitat around the trees (Broome, 2003; Gill, 1992b; Jactel et al., 2011), and several studies have shown that grazing and habitat management can reduce vole numbers (Andrews et al., 2000; reviewed in Jacob and Tkadlec, 2010). On former agricultural land where ground layer can be very rich (supporting high small rodent populations) use of herbicide for up to two years may be required to protect trees (Trout et al., 2004). Herbicide application may reduce numbers of voles but also have a behavioural impact if voles are wary of crossing bare ground due to predation risk, and hence even application in just the immediate vicinity of each tree may have a beneficial impact (Trout et al., 2004). Forestry Commission (2020a) recommend a 1 m diameter area for this purpose. A study of damage to sycamores found that 86% of unweeded or weeded to only 25 cm diameter were damaged by voles (and damage was more severe with higher mortality rates), compared to 75% for 50 cm diameter and 47% for 1 m diameter (reported in Davies and Pepper, 1993). Davies and Papper (1993) suggest that herbicide application or weeding via hoeing is preferable to mowing as a form of weed control around trees since the mown grass/vegetation competes for nutrients with the young trees. Mayle (1999) suggests that heavy grazing by livestock can reduce field vole populations (both via reducing of vegetation, but also trampling and disturbance to leaf litter) but does not seem to impact wood mice. It has been recorded that populations (and thus potential impacts) of several small rodents species, including yellow-necked mice, wood mice and bank voles may be reduced in the presence of heavy grazing by deer in woodlands (Flowerdew and Ellwood, 2001).

An alternative to herbicide application is the use of fabric or polythene mulch mats around planted trees. The effectiveness of these in the UK appears to have been limited, and they may simply provide shelter for voles (underneath them) which can then still damage bark and roots of the tree (Trout et al., 2004). Trout et al. (2004) report various trials of coating the underside of these mats with repellents. These included natural predator odour and chemical products Aaprotect, Renardine and Woebra, but at the time of publication the chemical products were not licensed for use against voles. Davies and Pepper (1993) also suggest that the inadvertent benefit of mulch mats to voles can be reduced by placing earth clods or other weights on the mats, but still suggest that vole guards are more effective. In a study on English oak in North America, mice and voles browsed about half as many saplings protected by fabric mats than unprotected saplings (reported in Dyderski et al., 2020). It has been also suggested that a layer of leaf litter can reduce predation of beech seeds and protect regeneration (Packham et al., 2012).

Because vole abundance is highly linked to grass abundance and cover, Springthorpe and Myhill (1994) indicate that a response to vole damage should be beating up (i.e. replacing any lost trees within a plantation, which help reach shading out of grass cover more quickly , see Taylor, 1943).

MODIFY TREE RESOURCE AVAILABILITY AND RESILIENCE TO DAMAGE

In European forests, Jactel et al. (2011) suggest that more diverse forest stands might reduce browsing by voles relative to monocultures, although this may depend on the species choice and relative palatability of each. There is some evidence that vole resistant tree varieties (probably differing in phenolic compounds) do exist for species such as Norway spruce, birch, lodgepole pine and Scots pine (Gill, 1992b). However, Rogers and Brambell (1958) suggest unpalatable planting is not effective against vole damage in the UK. Parratt and Jinks (2013) found that burying seeds deeper substantially reduced predation rates of a range of tree seeds relative to shallower burial. This is probably both because seeds are harder to detect (rodents locate seed primarily via odour: Engman, 2020), and because if detected they have a longer handling time, so rate of seed removal of a given species is reduced. In Scandinavia where vole damage can be extreme, timing of planting of new trees is modified so that trees are planted just after the population peak (Gill, 1992b). Jacob and Tkadlec (2010) suggest that development of predictive models of vole abundance could help in reducing damage in this way or targeting other protection methods most efficiently. Because vole damage appears to be higher in plantations than naturally regenerated forests, the use of natural regeneration as an afforestation method is sometimes used in Europe.

5.1.4 Lagomorphs: Rabbits and hares (Family Leporidae)

5.1.4.1 Background and relevant ecosystem services

There are three lagomorph species relevant to an English context, European rabbit and brown hare which are found throughout England, and mountain hare which is currently only found in the Peak District (Matthews et al., 2018). Both European rabbit and brown hare were thought to have been introduced to Great Britain about 950 and 2,000 years ago respectively (Baker, 2010; Lever, 2010). There was little information in the literature about ecosystem services provided by lagomorphs in woodland or treescapes. All three species have a greater association with open habitats, although Matthews et al. (2018) note them as important prey items for many native carnivores, and they also provide prey for raptors. Nevertheless, they can do substantial damage to trees via browsing and bark stripping. Forty-six grey sources and 45 scientific papers made some reference to rabbits or hares, 91 in total. Seventy-three referred to rabbits, 37 brown hares and 14 mountain hares.

5.1.4.2 Damage and impacts

Lagomorph damage has been widely reported in the literature and while we did not find as many sources relating to this taxon compared to deer, Putman and Moore (1998) report on a Forestry Commission survey of conifer plantings of 3 months to 3 years of age and reported that in England, rabbit damage was 3-10 times higher than that of deer. While these values do not reflect changes in rabbit and deer populations since the 1990s, they suggest a high degree of damage is possible. Outside of commercial conifer plantations, both rabbits and brown hares were observed to hinder re-establishment of hazel in coppices in Cambridgeshire (Cooke and Farrell, 2001). Putman and Moore (1998) also indicate rabbit and hare damage as a risk to coppice stool regrowth. Guidance on creation of new woodland via colonisation in England indicates that rabbit and hare (alongside exclusion of livestock and deer) need to be ‘closely controlled or eliminated’ in an area for natural colonisation of non-wooded areas to occur (Forestry Commission, 2021).

Rabbits can cause damage both via browsing or bark stripping. In a survey of 441 forest managers from 2021, ‘Keeping rabbits… from bark stripping’ was indicated as a factor promoting tree health (Bates et al., 2024). Rabbits are known to browse seedlings and young birch trees *Betula spp.* which can eliminate regeneration or stunt growth (Cameron, 1996), and Gill (1992b) suggests that almost any species of tree can be susceptible to browsing. Gill (1992b) notes that rabbit damage can occur very quickly and extensively on newly planted compartments and that presence of cover nearby is likely to increase risk of damage. Bark is stripped in patches of the main stem near ground level (typically 50 cm or lower: Trout and Brunt, 2014) and is typically on young trees but can also impact smooth-barked species such as beech. Gill (1992b) notes that damage is typically greatest in winter, particularly if there is prolonged snow cover. Pepper (1998) suggest that as well as beech, ash is particularly susceptible to bark damage from rabbits. There is some evidence that some fruit trees are more susceptible to bark stripping by rabbits than timber production trees (Gill, 1992b). Juniper can also be highly susceptible to browsing and basal bark stripping (Broome, 2003; Thomas et al., 2007), as can yew (Watt, 1926). Rabbit browsing (in the absence of deer) also had a substantial impact on growth of directly sown broadleaf tree seedlings (including oak, sycamore, maple, wild cherry, ash, pear and hazel) in Herefordshire (Willoughby et al., 2004).

Like damage from rabbit, Gill (1992b) notes that hare damage tends to be highest in winter and when there is snow lie. However, the author also notes that, unlike in other countries, bark stripping by hares in the UK appears to be rare. Hares are notable in damage occurring systematically by an individual along a row of planted trees (Gill, 1992b). Browsing of twigs up to 6 mm in diameter has been observed for mountain hares (Gill, 1992b; Rao, 2001) although a peak at 2 mm was observed by Rao (2001) in planted downy birch seedlings. In a study in the Czech Republic, Vacek et al. (2018) observed brown hare herbivory to be a major factor in very high mortality of naturally regenerated sycamore seedings, and the survival rate of seedlings browsed by hare was only 0.5%. In another study in the Czech Republic, hare damage (species not specified) was more likely to cause growth deformation rather than mortality of saplings (Krojerova-Prokesova et al., 2018). Forest Research (2024b) indicate that most hare damage occurs in the establishment and thicket stages of tree growth.

It has been noted that in upland areas of Britain, tree regeneration can be significantly impacted by hares, although there may be a preference for willow and rowan over birch (summarised in: Mitchell and Kirby, 1990). Based on habitat selection analyses from tagged animals, Rao (2003) observed no selection by mountain hares in Scotland for moorland with trees (principally naturally regenerated and planted downy birch and Scots pine of < 1.2 m), even in winter. The author concluded that tree damage risk by the species will likely be a function of high numbers rather than direct selection. However, Rao (2001) found that while browsing on naturally regenerated trees was not predicted by variables such as hare abundance, tree species or tree density, browsing of planted downy birch was high in winter and where tree density and hare density were higher, as well as where heather was shorter. In a review of European forest hazards Jactel et al. (2011) indicated that shrubs provide shelters for hares which may increase risk of damage to newly established conifer seedlings.

5.1.4.3 Identification of damage

Van Uytvanck et al. (2008) note that rabbit damage can be distinguished from large herbivores because they "cut seedling stems with a typical smooth, sloping sectional plane". However, on Scots pine in Norfolk it was reported that that deer and lagomorph (rabbit and hare) leader damage could not always be distinguished reliably (Zini et al., 2022). Springthorpe and Myhill (1994) provide a clear illustration of leader browsing differences between rabbit and hare (clean often diagonal cut, with separated shoot often left lying) and deer (often torn and ragged cut, and the removed shoot itself is always consumed). This contradicts somewhat with recent guidance that suggests rabbits often consume the cut stem, whereas hares do not (Forestry Commission, 2023). Hare and rabbit browsing can be hard to distinguish but, as discussed below, rabbits will tend to browse progressively away from cover, whereas hares will tend to browse sequentially along a line of trees (Springthorpe and Myhill, 1994). Bark damage by rabbits and hares might be confused with that of deer but will tend to be lower down (typically <60 cm) and not generally above any grass mat, unlike bark damage of field voles. Recent guidance for England suggests that the height of damage by hares can reach 70 cm and rabbits 54 cm (Forestry Commission, 2023). This concurs with guidance from Scotland (Andrews et al., 2000). Width of teeth marks has been reported as < 3.2mm (although Forestry Commission, 2023 report a range 3-4 mm for rabbits) and frequently diagonal to the tree stem and may be seen in areas inaccessible to deer (Springthorpe and Myhill, 1994). Springthorpe and Myhill (1994) note that the upper incisors of rabbits and hares are deeply furrowed so a clear tooth mark may show four furrows on the tree, not two. Rabbit and hare damage to large, smooth-barked trees may occasionally resemble that of grey squirrels but would occur in the winter or early spring so can be distinguished from later damage caused by grey squirrels (Mayle et al., 2007; Springthorpe and Myhill, 1994). Andrews et al. (2000) also report this pattern of rabbit and hare damage to trees tending to be greatest in winter and early spring.

It should be noted that rabbit and vole damage can be confused (Harmer, 1995). Characteristics of rabbit damage can include small trees being ‘cut through’ at the root collar and evidence of shoots and buds left lying by the tree (Gill, 1992b), and they can graze tree/shrub species seedlings down to the ground (Grubb et al., 1999). Putman et al. (2011) suggest that any browsing damage above 60 cm cannot be attributed to rabbits. Damage to yew (bark stripping and browsing) have been reported at 3-4 feet (91-122 cm) (Watt, 1926) although this was in winter and snow lie may have aided higher browsing. This is backed up by Springthorpe and Myhill (1994) who suggest rabbit (and hare) browsing can reach 1.5 m in deep snow lie. This concurs with more recent guidance which suggest a maximum height of 54 cm for rabbit damage, but higher in snow (Forestry Commission, 2023). Trout and Brunt (2014) indicate that trees damaged by rabbits are typically younger (1-10 years old) although trees with diameters up to 1 m have been damaged by rabbits. Forest Research (2024b) suggest damage can primarily occur from seedling to pole stage and they also suggest that rabbit burrows can assist windthrow and also may provide weak points for wild boar to dig under fencing.

5.1.4.4 Protection against damage

Tree protection methods identified in the literature for lagomorphs are shown in Figure 11, with details and evidence base synthesised below.

REDUCE TREE RESOURCE DEMAND

Rabbit eradication has been successful on some British islands (Mill et al., 2020) but the closest to national scale eradication was the use of the Myxoma virus as a biological control agent. This was chiefly from the 1950s and it was highly effective in reducing numbers, although both genetic resistance and reduction in virulence have been observed (Baker, 2010). The near complete collapse and localised eradication of the rabbit population following introduction of myxomatosis (estimated populations fell from c. 100M to 1M in about six years: Lever, 2010) was apparently associated with both a large increase in woodland regeneration and tree/scrub colonisation of grasslands (Gill, 1992b). Macmillan and Phillip (2008) report a study from the 1980s putting the cost of rabbit control in the UK at £5M PA (about £15M today), although this will include control for agricultural protection as well as tree protection. White et al. (2011) reported rabbit control elements of ranger costs of Forest Enterprise (the former body responsible for managing and promoting publicly owned forests in the UK) in the mid-1990s to be >£0.5M (about £1M today). The Forest Plastics Working Group (2022) indicate that provision of owl/kestrel nest boxes can reduce incidence of rabbit damage although do not provide evidence for this.

Rabbit protection has relied heavily on lethal control. A range of approaches are still used including netting and shooting, with and without the use of ferrets *Mustela furo* to chase rabbits out of burrows, plus the use of snares and various forms of lethal traps (Baker, 2010). In addition, gassing/fumigants have been used which can be effective for burrow dwelling animals, and Hodge and Pepper (1998) indicate that this is the most effective method of rabbit control, albeit requiring extensive training. According to the Wildlife Rangers Handbook (Springthorpe and Myhill, 1994) and Forestry Commission guidance (Pepper, 1998) rabbit control is often focussed in autumn and winter to reduce numbers before the regrowth begins in spring, and winter monitoring for bark stripping and control is also recommended. The Handbook also recommends mowing of grass rides in September to facilitate shooting over the winter (Springthorpe and Myhill, 1994). Springthorpe and Myhill (1994) note that it is illegal to use spring traps for rabbits unless within their burrows. Current Natural England Guidance (Natural England, 2015) indicates that traps and snares, gas, ferreting and shooting are all legal methods subject to certain legal restrictions and best practice. Hodge and Pepper (1998) and Pepper (1998) provide some comparative assessment of these methods in terms of effectiveness and operational/capital costs.

Brown hare were frequently hunted using dogs but this practice is now illegal (Baker, 2010). Shooting (with the use of retrieval dogs) is the recommended control method and a code of practice is available (British Association for Shooting and Conservation, 2024a). It is noted in Springthorpe and Myhill (1994) that it is illegal to set spring traps for hares.

MODIFY ACCESS TO TREE RESOURCES BUT NOT AVAILABILITY

One of the principal non-lethal methods of tree protection from rabbits has been via fencing (Buckley et al., 1997). A fence height of 1 m has been considered high enough to exclude rabbits and brown hares (Cooke and Farrell, 2001). However, Trout and Pepper (2006) recommend minimum heights of 0.9 m and 1.05 m for rabbits and hares respectively (although they note the data for hares is sparse) and Trout and Brunt (2014) later recommend 1.05 and 1.20 m. To exclude lagomorphs, a fence must at least have a dense mesh near the ground (Prach et al., 1996). Trout and Pepper (2006) give details about aspects such as size and spacing of woodwork for fencing for rabbits and hares, and they note that typical rabbit fencing consists of hexagonal twisted galvanised steel. Importantly, Trout and Pepper (2006) indicate that a turned base of at least 15 cm is essential and a top 45° overhang of 15 cm is desirable (both turned outwards away from the area being fenced). However, since rabbits can climb them, the wooden struts should not be on the outside of the fence. The turned base should be pegged or covered with sods of earth (Trout and Pepper, 2006) or buried (Pepper, 1998). Rabbit fencing should be kept clear of vegetation (such as bracken) so that any burrowing activity can be identified (Trout and Pepper, 2006). Harmer (1995) reviewed several studies that had used fencing to exclude rabbits and found an improvement in survival, 'quality' and number of oak seedlings. Rabbit proof fencing used in the Czech Republic had a hexagonal wire mesh 13x13 mm with 1mm diameter, with resistance to corrosion provided by a PVC coating (Marada et al., 2019). However, Trout and Pepper (2006) suggest 1mm is too thin but suggest a wider mesh (31-36 mm) is sufficient, albeit it may allow very small rabbits to pass through. For Countryside Stewardship, 31 mm mesh is required (Rural Payments Agency and Natural England, 2024). Forest Research (2024c) indicate that there are fewer options for tree protection for hare compared to rabbit, with tree guards and fencing suggested as the options available. In an exclusion experiment specifically for hares *Lepus spp.* in Finland, a mesh size of 5 cm has been used (reported in Agra et al., 2016).

Rabbit fencing may require badger gates where it crosses badger paths (Springthorpe and Myhill, 1994) and in future mitigation (such as stiles) may be required for wildcats *Felis sylvestris* which can get caught in fine mesh fencing (Herrman, 2005). It is noted by Trout and Pepper (2006) that the choice of a pipe under a fence (which is sometimes used as an alternative to badger gates) will prevent muntjac passing but not rabbits. Where a fence must be crossed by a road, but the road cannot be gated or gridded, so-called 'wings' of the fencing can be constructed along the road sides, reducing the availability of cover for any animals that might otherwise try to cross, and for rabbits Trout and Pepper (2006) recommend they are at least 50m long (see that guide for a diagram). Rogers (1965) provide some illustrations of modifications that may be made to a rabbit fence to deal with issues such as water courses crossing the fence line or where it goes past higher objects such as stumps that could allow rabbits to climb over.

When erecting rabbit fencing, Springthorpe and Myhill (1994) indicate that it is critical that piles of brash or other cover are cleared and rabbits are eliminated before a rabbit fence is constructed, and then rabbits inside the fence are removed via lethal control. This process needs to be repeated regularly in case any rabbits remain in the area or have burrowed in, and particularly if snow drift may have allowed fence incursion. Pepper (1998) suggest that in high density rabbit areas this process may require some internal subdividing fencing. Trout et al. (2004) also suggest that retained brash or windrows following felling can increase rabbit population density and Andrews et al. (2000) also report this for both rabbits and hares in upland landscapes.

Trout and Pepper (2006) note that temporary rabbit fencing can be used around coppiced areas or short rotation biomass crops, and they give recommended specifications for reusable materials for doing so. Guidance from the 1990s (Hodge and Pepper, 1998), and some current resources (Forest Research, 2024c) suggest that electric fencing was not a viable tree protection method against rabbits. Nevertheless, electrified plastic netting (containing fine wires) can be used for temporary exclusion of rabbits (although it is noted this could be up to 7 years if well maintained) and these might be useful for short-term exclusion (Trout and Pepper, 2006). Where wire strands are being used for electric fencing, Trout and Pepper (2006) recommend size live wires spaced between 5 and 10 cm apart between 10 and 40 cm high, with an earth at 5 cm. It is noted that electric fencing must be live when first constructed or rabbits are more likely to cross even when later live (Trout and Pepper, 2006).

A small amount (3% of stems) of brown hare browsing to young ash stems inside electric fencing (five equally spaced strands with the highest at 95cm) was found in Cambridgeshire (Cooke and Lakhani, 1996). Springthorpe and Myhill (1994) note that rabbit fencing is effective against hares, but its cost is rarely justified by hares alone as they tend to create less tree damage than rabbits, particularly as they do reach densities close to those of rabbits.

Prior to the introduction of plastic tree tubes/shelters, wire mesh individual tree protection was used where fencing was impractical (Pepper, 1976). Early tree tubes/shelters used to protect against rabbit grazing were produced at a height of 60 cm (Tuley, 1985). Broome (2003) reports a height of 60 cm as also effective against hare browsing on juniper although this is not provided with evidence. However, Kerr (1995a) reported that tree shelters of 75cm were effective at 58% of sites where hares were the tallest herbivore in a survey of 193 sites across England. In an area of mixed deer, livestock and rabbits Broome (2003) reports juniper in 60 cm high 4x4 cm square tree shelters as having higher growth rates than those outside shelters although survival was not significantly higher, albeit the shelters collapsed or disintegrated after five years. Kerr (1995b) reported a 100% effectiveness rate of 60 cm tall tree shelters at sites where rabbit was the tallest herbivore in a survey of 193 sites across England. In some cases, tree tubes have been shown to be more effective than rabbit fencing although details of the fencing the and condition are not provided (Potter, 1991). It has been noted that rabbits may eat cable ties used to fix tree tubes (Yorkshire Dales Millenium Trust, 2024). It has also been observed that a type of tree guard made from 100% recycled material and fully compostable (the “Ezee Tree”) was partially damaged by rabbit nibbling although it is unclear if the extent of this damage reduces their effectiveness (Yorkshire Dales Millenium Trust, 2024).

Armstrong and Robertson (2013) also suggest that spiral vole guards might be effective against rabbit damage but do not provide evidence. Pepper (1998) suggests spiral guards are an alternative to tree tubes for feathered trees (trees where branches begin at the base as opposed to standard trees), but they caution that the spirals most be wound between branches and no gaps left between spirals as rabbits can gnaw bark through a gap of just 5 mm.

Thompson (1953) and Armour (1963) review a range of chemical repellents for rabbits and hares on forest and fruit trees, some of which were claimed to be able to protect trees with a single application. These included naturally derived substances such as coal tar and wood tar, and a substantial number of chemical formulations. However, it is unclear whether such repellents would be legal under current legislation. ‘Aaprotect’ is effective and approved and can protect against winter rabbit browsing and bark stripping but can only be used during the dormant season (from mid-November) due to its phytotoxicity (H. Armstrong et al., 2003; Hodge and Pepper, 1998). It has been shown to be 95% effective in reducing browsing of deer and rabbits (Pepper, 1978) and can be applied by painting, dipping or spraying (Pepper, 1976). Aaprotect is also claimed to be effective against hares (Pepper, 1978)

While the use of herbicides and scarification to reduce grazing availability for rabbits and hares (and thus decrease use of an area with young trees) appears to have some success on other lagomorph species globally, Gill (1992b) suggests there is not strong evidence for this for UK species. Trout and Brunt (2014) also suggest that mowing to maintain short vegetation around trees may make rabbits more wary of approaching them if far from cover.

In an experimental study in the Netherlands, Bakker et al. (2004) found that rabbits suppressed the survival and growth of 1-year-old transplanted English oak *Quercus robur* seedlings but also that blackthorn *Prunus spinosa* scrub did not offer associational resistance to the oak seedlings. They suggested that this was both because the blackthorn has fewer spines lower down and that rabbits may actively suppress its clonal expansion via browsing. Indeed, the latter effect may mean associational resistance from large herbivores is less effective in areas with rabbits (reviewed extensively in Armstrong and Robertson, 2013). In an experiment in Belgium, planted seedlings of English oak and ash had their highest mortality due to rabbit grazing when planted among bramble patches (Jan Van Uytvanck et al., 2008). They hypothesised that this was due to the cover these areas provide for rabbits and potentially poorer growing conditions with reduced light. Thus, rabbits may reduce any benefit cause by associational resistance. Rabbit browsing (in the absence of deer) also had a substantial impact on growth of seedlings of directly sown broadleaved tree species (including oak, sycamore, maple, wild cherry, ash, pear and hazel) in Herefordshire (Willoughby et al., 2004). Although the effect of associational resistance of thorny scrub does not seem to impact rabbit damage (discussed above), cut branches of gorse packed around the base of young trees have been tried but its effectiveness is not reported (Armstrong and Robertson, 2013).

MODIFY TREE RESOURCE AVAILABILITY AND RESILIENCE TO DAMAGE

In European forests, Jactel et al. (2011) suggest that more diverse forest stands might reduce browsing by hares relative to monocultures, although this may depend on the species choice and relative palatability of each.

5.1.5 Pigs and wild boar (Family Suidae)

5.1.5.1 Background and ecosystem services

Wild boar *Sus scrofa* are among the most widely distributed mammals in the world (Oliver, 1993). The species' natural range encompasses Eurasia and northern Africa; however, introduced populations are now present on every continent except Antarctica (Long, 2003). Although once native to the United Kingdom, wild boar became extinct in the wild by the 17th century due to habitat loss and overhunting (Albarella, 2010; Yalden, 2010). In recent decades, however, several feral wild boar populations have become established in southern England, following escapes from farms and deliberate releases. The largest of these populations is located in the Forest of Dean, Gloucestershire, where estimates indicated a peak of 1635 individuals in 2018/19 (Forestry England, 2024). Smaller populations can also be found in parts of Kent, Sussex, Herefordshire, Devon, and Dorset (Wilson, 2013, 2003), as well as localised areas in Wales and Scotland (Forest Research, 2024b). Numbers of individuals within these feral wild boar populations are reportedly increasing, mirroring trends observed in many other parts of Europe, where the species is now frequently regarded as a pest (Massei et al., 2015; Tack, 2018). With the highest reproductive rates among ungulates, wild boar are capable of annual increases in population size of up to 150% (Massei and Genov, 2004).

The success of wild boar can be attributed, in part, to their high behavioural plasticity. Wild boar inhabit a number of habitat types, ranging from semi-arid deserts to marshes, forests, alpine grasslands, and urban and agricultural environments, but they prefer those that offer high-energy food and cover from predators (and hunters) (Massei et al., 2015; Tack, 2018). The species is also likely tolerant to a variety of climate conditions, reflected in their extensive natural and introduced geographic range (Oliver, 1993).

Wild boar are generalist omnivores, with their diet largely determined by the availability of different plant, fungi, and animal food types, which can vary with season, year, and location (Schley and Roper, 2003). Typically, more than 90% of their diet is dominated by leaf matter, berries, nuts, seeds and mast, and roots and rhizomes (Schley and Roper, 2003). Energy-rich foods, such as acorns and beech mast, represent their preferred natural diet. When natural food sources become scare, wild boar may also feed on agricultural crops, such as potato, rapeseed, sunflower, and maize (Massei and Genov, 2004; Schley and Roper, 2003; Tack, 2018). In England, most of the agricultural damage caused by wild boar has been to maize crops, and maize is frequently used as an artificial bait by hunters to attract wild boar into safe shooting areas (British Association for Shooting and Conservation, 2024b). A recent study using DNA metabarcoding of faecal samples of wild boar harvested by hunters during autumn and winter in Norway revealed a high degree of diet variability among individuals, and between sexes (Mysterud et al., 2024). Male wild boar were found to consume 50% more ‘non-natural’ food sources (i.e., agricultural crops) than females. The factors influencing the amount of crop damage can include the local density of wild boar density, the availability of natural food sources, and the proximity of agricultural fields to woodlands or forested areas (Massei and Genov, 2004).

While foraging, wild boar engage in a behaviour known as ‘rooting’, which involves overturning extensive areas of soil to access food located in leaf litter or beneath the ground. Rooting can provide various beneficial ecosystem services, such as reducing rank vegetation and bracken cover and promoting new plant growth by uncovering dormant seeds in the soil (Scottish Forestry, 2024; Sims, 2006). However, wild boar frequently re-root the same general areas within their home range (c. 400 – 15000 hectares; Massei and Genov, 2004), which can lead to significant disturbance (Goulding, 2003). Excessive rooting has been shown to negatively impact ecosystems by mixing soil horizons, reducing vegetation cover and leaf litter, increasing nitrate concentrations and soil respiration, and decreasing the abundance of soil arthropods (Mohr et al., 2005; Sims, 2006; Singer et al., 1984).

5.1.5.2 Damage and impacts

Rooting by wild boar can damage trees at all stages of growth, from seedlings and establishing trees to thickets and pole-stage growth (Forest Research, 2024b). Mature trees are less frequently affected by rooting, but they nevertheless remain vulnerable to damage. For example, wild boar may rub against mature trees, stripping or damaging their bark (Forest Research, 2024a; Scottish Forestry, 2024). If the bark is removed from the entire circumference of the tree — a process known as ‘girdling’ — the flow of nutrients and water from the roots can be disrupted. This can lead to nutrient deficiencies, increased susceptibility to diseases, stunted growth, and ultimately the death of the tree.

Rooting can also damage fences and lead to a reduction of valued ground flora, such as pignut *Conopodium majus* and bluebell *Hyacinthoides non-scripta* (Harmer et al., 2011). In areas with high densities of wild boar, rooting can completely remove woodland ground flora, although the impact of rooting can be uneven and unpredictable (Goulding, 2003; Harmer et al., 2011; Scottish Forestry, 2024; Sims, 2006).

5.1.5.3 Identification of damage

The most common indication of damage caused by wild boar is the disturbance of soil and vegetation due to rooting. Wild boar root to an average depth of 5 to 15 cm (Massei and Genov, 2004). The size of individual rooted patches, and the large snout marks often visible at the edges of the patches (Wilson, 2003). Wild boar hair caught in fences or damaged fences can also be a sign of their presence. The hair of wild boar has distinctive whitish split ends, making it easy to distinguish from that of deer and other wildlife (The Deer Initiative, 2009).

5.1.5.4 Protection against damage

Tree protection methods identified in the literature for wild boar are shown in Figure 12, with details and evidence base synthesised below.

MONITORING BEFORE INTERVENTION

Since 2013, the feral wild boar population in the Forest of Dean has been monitored using an annual "distance sampling/thermal imaging" survey (Forestry England, 2024). Originally developed by Forest Research to track deer populations in woodlands and forests, this technique has enabled Forestry England to effectively monitor changes in wild boar population dynamics over time (Forestry England, 2024).

REDUCE TREE RESOURCE DEMAND

In Europe, wild boar are vulnerable to predation by species such as the Eurasian lynx *Lynx lynx*, brown bear *Ursus arctos*, and especially the Eurasian wolf *Canis lupus lupus* (Tack, 2018). Although all three of these predators were once native to the United Kingdom, they were hunted to extinction by the 17th century. As a result, feral wild boar populations in the United Kingdom lack any natural predators (Forestry England, 2024). Although predation may slow the growth of wild boar populations, their high reproductive rates enable them to recover quickly; hence, predation alone is unlikely to be enough to control wild boar numbers.

Shooting is the most common hunting method used to control wild boar populations. In the United Kingdom, The Deer Initiative (2009) and BASC (2024a) provide guidance on how to shoot wild boar humanely and safely, including recommendations on appropriate firearms. However, a recent study by Massei et al. (2015) examined the correlation between the number of hunters and wild boar population sizes across Europe and concluded that current hunting levels were insufficient to limit wild boar population growth (see also Keuling et al. 2013). Supporting Massei et al. (2015), a report by Green (2023) stated that “it is widely accepted that methods used to control wild deer — namely daytime stalking, high-seat shooting, and night shooting — are not effective in managing wild boar populations”.

Wild boar can be trapped using various methods, including drop nets, cage traps, panel traps, and corral traps. In a pilot study conducted in Scotland for Nature Scot, Green (2023) tested the effectiveness of capturing and culling groups of wild boar using drop nets and corral traps at pre-baited sites. Based on the findings of Green (2023), and guidance by The Deer Initiative (2009), it is arguable that these techniques are only effective for research purposes – such as population monitoring – or for dealing with specific individual problem animals or small groups, rather than for controlling populations.

The use of contraception for controlling wild boar populations is the subject of on-going research (Massei, 2023; Massei et al., 2012, 2008) but such studies on feral wild boar populations in the United Kingdom are only at the pilot stage (Quy et al., 2014) and limited to small sample sizes. In a captive study of wild boar in the United Kingdom, Massei et al. (2008) found that 11 out of 12 sows treated with a Gonadotrophin Releasing Hormone (GnRH) immuno-contraceptive vaccine remained infertile for at least 4 to 6 years following a single injection. There were no reported adverse effects on physiology or behaviour.

Chemical deterrents for use in the United Kingdom must be approved under the Control of Pesticides Regulations (1986). There are none currently approved or that have been tested specifically as wild boar repellents. Diversionary feeding should not be regarded as a solution to localised wild boar damage, and in the long term it could potentially support an increase in numbers and range (The Deer Initiative, 2009).

MODIFY ACCESS TO TREE RESOURCES BUT NOT AVAILABILITY

The Deer Initiative (2009) provides a best practice guide of fencing standards aimed at preventing both the escape of captive wild boar and the introduction of feral wild boar into areas where their presence is undesirable. Wild boar are large, powerful animals capable of digging and jumping. Consequently, fences must be strong enough to withstand the impact of wild boar running into them while being elastic enough to avoid causing injury. Wild boar are adept at breaching many standard fencing designs.

In brief, The Deer Initiative (2009) recommends that fences should be at least 1.5 m high – and ideally 1.8 m high – and buried 0.5 m below ground level and constructed from 2.5 mm diameter high-tensile netting. Electric fencing can be used, although it may be more appropriate for containing captive wild boar that have been trained to avoid the shock. Electrified plastic mesh fencing is not recommended as wild boar can become entangled in it (The Deer Initiative, 2009). The use of barbed wire is not appropriate except as a single strand at ground level where wild boar digging under fences are a problem. Gates in fences can be an obvious weak point. The standard ‘5 bar gate’ is not sufficiently high to exclude wild boar without the use of electrified stand-off wires or additional wire mesh.

5.1.6 Deer (Family Cervidae)

5.1.6.1 Background and ecosystem services

There are six deer species present in England, only two of which (red and roe deer) are native, with fallow deer introduced some 1,000 years ago, and sika, muntjac and Chinese water deer within the last century or two (Woodland Trust, 2013b). In some regions of the country, all six deer species can be found (e.g. Wildlife Trust for Beds, Cambs and Northants, 2024). Most are associated with woodland habitats to varying degrees, apart from Chinese water deer which is associated with wetlands (Matthews et al., 2018). As a whole, deer populations in the UK are growing and are recognised as a key national problem for woodland creation and biodiversity (The UK Government, 2021). They lack natural predators to regulate numbers (Woodland Trust and Hotchkiss, 2020).

Because of their large size, high densities, diversity of species and strong association with woodland ecosystems the impacts of deer are widely discussed in the literature. Beyond impact on trees, deer can have a big impact on the non-tree vegetation in a woodland because of relative palatability/preference of different species (The Deer Initiative, 2024). Exposure studies have demonstrated that exclusion of deer where they were previously at relatively high densities can substantially increase vegetative cover within a few growing seasons (Game and Wildlife Conservation Trust, 2024). However, where a mixed suite of large herbivores is excluded (deer and livestock, for example) it is more challenging to tease apart the relative impacts of different species (e.g. Hasstedt and Annighoefer, 2020; Murphy et al., 2022). Using terrestrial laser scanning to examine three-dimensional woodland structure, Eichhorn et al. (2017) found that lowland woods in England and Wales with deer densities > 10 per km^2 had 68% lower density of understory foliage in the 0.5-2 m height zone than woods with c. 1 deer per km^2. As discussed with rabbits complete lack of grazing deer in woodlands may lead to domination by bramble which shades out other woodland ground flora (Watkinson et al., 2001).

Via their impact on vegetation deer and other large herbivores can impact biodiversity of other taxonomic groups. In a global meta-analysis of studies that had looked at associations between large herbivores and other groups, there were negative associations with richness of small mammals, arthropods overall, spiders, true bugs and lepidopterans (Foster et al., 2014). This study concluded that that the primary mechanistic reasons provided by reviewed studies related to changes in vegetation biomass and structure. Several studies in England have found negative impacts of deer on other taxa. In a deer exclosure experiment over four years in a coppiced wood in eastern England, Holt et al. (2014) found positive associations between deer exclusion and shrub-foraging and ground-foraging birds, but no effect on canopy-feeding guild. They attributed effects partly to the benefits of improved thermal cover in winter and nesting cover in spring, and overall improved cover from predators. For one understory browsing species studied in more detail at the same study sites, the black cap *Sylvia atricapilla*, Holt et al. (2013) found that densities and body condition were higher in unbrowsed plots. Males settled on territories earlier, suggesting higher habitat quality for the species. Similar results to Holt et al. (2014) were found by Newson et al. (2012) on a national scale across England. Estimated deer densities were modelled against change in bird abundance over time. Five woodland bird species associated with dense understory vegetation saw declines associated with higher deer numbers (nightingale *Luscinia megarynchos*, song thrush *Turdus philomelos*, Chiffchaff *Phylloscopus collybita*, willow warbler *Phylloscopus* *trochilus*, and willow tit *Poecile montanus*) and no species displayed a positive impact. However, two species not associated with dense understory vegetation did increase in association with higher roe deer numbers by a small amount (blue tit *Cyanistes caeruleus* and robin *Erithacus rubecula*). In north-central France in a study investigating deer and wild boar and densities, vegetation structure and bird communities in mature forest stands did not find any negative association between higher deer browsing pressure and richness or abundance of birds (Baltzinger et al., 2016). The authors suggest that impacts may be more likely in younger or regenerating forest. There is some evidence from North America that long-term high deer densities can lead to increased soil compaction. and reduced soil prokaryotic diversity (Maillard et al., 2021) although this has not been tested in the UK to our knowledge.

Although high deer impacts can have a profound negative impact on structural diversity and ground flora species richness in woodlands, complete lack of browsing can also lead to a reduction in habitat heterogeneity due to loss of open woodland areas. Low levels of deer browsing are likely to be beneficial overall for biodiversity relative to high or no browsing (Armstrong et al., 2023; Woodland Trust, 2022). In a study in Atlantic oakwoods in Scotland, Moore and Crawley (2014) found that deer exclosures increased shading which was associated with a reduction in diversity of cryptogram diversity/richness. It has also been argued that some low level of browsing promotes diversity of tree growth forms which will improve woodland structural diversity (Woodland Trust, 2022). Addressing the question "are there too many or too few large herbivores in our woods?", Kirby (2017) argued that both can be true depending on context and the objectives for the woodland. In some cases, deer can have positive associations with other elements of biodiversity. For example, in mixed montane forest in southern Germany roe deer were positively correlated with carabid beetle richness (Cordeiro Pereira et al., 2024). Broughton et al. (2022) also argue that the elongated tree cavities often created as a result of bark stripping by deer (and European bison) and subsequent stem rot by deer are likely to be an important resource for cavity nesting birds in woodland, such as tits and flycatchers. As with other large mammals, deer are seed dispersers (Woodland Trust and Hotchkiss, 2020), and small seeds caught on the fur or hooves of deer can be dispersed (The Woodland Trust, 2019).

5.1.6.2 Damage and impacts

Browsing refers to feeding on buds, shoots and foliage of plants, including trees (Forest Research, 2024b). In a survey of 695 woodland managers, deer browsing was frequently mentioned a key risk and cause for uncertainty for regeneration of conifers and broadleaf woodland (Royal Forestry Society, 2021b). Because of deer browsing levels and preferences, discussed below, there is a concern that woodlands are becoming less diverse in their tree species composition (Woodland Trust, 2020). In some cases, preferential browsing can favour regeneration of non-native species such as Douglas fir over native species (The Woodland Trust, 2019). In a study in northern France, it was shown via an experimental trial with deer exclusion plots that deer browsing was leading to a 'substitution' of silver fir with Norway spruce over time due to preferential browsing, and thus reduced sapling density and height, on the former.

Deer damage can depend on species due to factors such as size, differences in behaviour and dietary differences. For example, meta-analysis of dietary studies across Europe show that roe deer are primarily browsers in both the growing season and the winter, red deer can perform a more diverse mix of browsing and grazing, with (with increased grazing in the growing season) and fallow deer are primarily grazers in the growing season although increase their browsing in winter (Spitzer et al., 2020). Across 10 sites in Northern Europe from Norway to Russia, Angelstam et al. (2017) found that a large herbivore index (which was based on red deer, roe deer and moose) had a significant positive relationship with browsing damage.

In a survey of 441 UK forest managers from 2021, those who managed oaks identified bark damage (including that of deer) as the biggest threat compared to aspects such as disease, drought and environmental change (Bates et al., 2024). Deer can cause damage on trees at growth stages from establishment through to mature, although muntjac deer damage tends to be on younger trees (Forest Research, 2024b). All deer species will browse seedings.

At high browsing levels there will be no natural regeneration and only saplings with their leader above browsing height will survive (Armstrong et al., 2023). Leader shoot browsing can be compensated by trees through new buds, new shoots or epicormic shots, and this compensation might be particularly stronger where growing conditions are better (Kupferschmid et al., 2013).

Some species appear to be less vulnerable to deer browsing as seedling, for example birch (The Deer Initiative, 2024) and beech (Packham et al., 2012) and palatability rankings or assessments for large herbivores and deer have been produced (Hatchkiss and Herbert, 2022; Scottish Forestry, 2024). However, palatability is not the only factor in terms of survival, and some species tend to be more resilient to browsing. For example, conifers tend to me more vulnerable to mortality due to deer (or other herbivore) damage as they store more of their nutrients in their leaves relative to broadleaf species (Scottish Forestry, 2024). Additionally, when deer pressure is high, even less palatable species such as spruce and beech are browsed (The Woodland Trust, 2019).

Critical to commercial forests can be browsing of the leader (dominant) shoot, the browsing of which can reduce forest crop harvest value due to deformed growth patterns this causes. In a study of young planted pine stands in Thetford Forest, Zini et al. (2022) were interested in relative damage caused by a mixed deer (and hare) species assemblage, and they found that trees were most vulnerable to leader damage when 1 year old and this risk reduced with age, and that the biggest impact came from fallow and red deer, with roe and muntjac (and hares) not correlating to degree of damage.

Where it is intense, browsing can cause coppice stools to die off (The Deer Initiative, 2024).

Fraying is a physical impact of antlers on trees and is used both to remove velvet from antlers and as a territorial marker (Forest Research, 2024b). Stem breaking can occur due to fraying but is also used by deer to help browse shoots that are out of normal reach (The Deer Initiative, 2024), although apparently is not carried out by roe deer (Armstrong et al., 2023). At high levels, stem breakage can reduce height growth of a substantial proportion of saplings (Armstrong et al., 2023).

Fraying is most common between March and May (Forestry Commission, 2023). Fraying can become 'thrashing' in the larger deer species which can cause breakage of branches and stems, as well the typical bark removal (The Deer Initiative, 2024). Muntjac deer also fray bark using their tusks (Forestry Commission, 2023). Fraying may be more common on saplings (up to 2 m) than on more mature trees (Armstrong et al., 2023).

Bark stripping itself is normally associated with the larger deer species (red, sika and fallow) (The Deer Initiative, 2024). Bark can form a substantial part of the diet of deer, particularly in winter when foliage food sources are diminished (Konopka et al., 2023), although it is observed year round (Forestry Commission, 2023). Bark stripping may be more common on smoother-barked species and trees >2 m (Armstrong et al., 2023). Cukor et al. (2022) found in Scots pine that on average it first occurred at about 18 years. At high levels, bark stripping by deer can lead to mortality of the tree (Armstrong et al., 2023) or stem breakage (Cukor et al., 2022). It also can lead to sublethal effects such as stem rot and structural defects that reduce timber quality and value. Although some species such as Scots pine may be relatively resilient in terms of growth, if the damage is not extensive (Cukor et al., 2022), other species such as Norway spruce have shown substantial impacts of stem rot on tree growth after deer damage, which are still significant even with less extensive bark damage (Cukor et al., 2019; Vacek et al., 2020). Fallow deer are known for pulling newly planted trees out of the ground (Forestry Commission, 2023).

The overall cost of deer damage can be very large and can reduce final timber crop values by up to 30-50% (reported in Forestry Commission, 2023) and is argued to 'nullify' economic returns from forestry in some cases (Mill et al., 2020). Estimating the overall cost of deer damage to forests on a national scale is very challenging and, in some cases, not possible due to lack of, or age of, data required to do so. Putman (2012) review evidence of economic damage to forests in Scotland but state that a single headline figure of the costs would be meaningless.

A major issue of deer damage is preventing regeneration and thus recruitment of mature trees. This effect may not have immediate impacts on spatial extent of woodland cover but over a long term where recruitment does not replace death of mature trees this can lead to loss of woodland, as has been suggested is happening in upland areas (Holl and Armstrong, 2014). This is supported by modelling based on field data by Tanentzap et al. (2013) who found that red deer densities in the Scottish Highlands are likely to reduce transition from younger to older (>3 m) growth stages. Deer can also predate seeds of tree species, for example acorns are consumed by red, roe and fallow deer (Dyderski et al., 2020), and in continuous cover forest systems, this can have substantial impacts on seed reserves (Putman, 2012).

5.1.6.3 Identification of damage

Deer damage at moderate deer densities will often create a 'browse line' in a wood about at 50-180 cm in height (depending on species present) (The Deer Initiative, 2024). Lower deer damage can overlap with that of rabbits or hares, particularly as these species can browse higher that might be expected by standing on their hind legs or when there is snow lie (The Deer Initiative, 2024). Fraying by the smaller deer species can also be confused with rabbit or squirrel damage, both of which can strip bark lower down tree stems, as can bark stripping using the teeth. However, in the latter case the larger teeth width of deer may aid identification (The Deer Initiative, 2024).

Deer lack teeth in the front upper jaw and their lower incisors bite against the roof of the mouth, which creates a distinctive ragged edge to damage twigs, which is different to the sharp, knife-like and often diagonal cut of rabbits and hares (Forest Research, 2024b). It can be challenging to distinguish deer and sheep damage, but sheep wool will typically be left where sheep are browsing (Forest Research, 2024b). Armstrong et al. (2023) give illustration indicating how to identify if browsing is from the current year or older.

Browsing height can help identify the species involved (or at least rule out some species), with roe deer browsing to 1.1 m, muntjac to 1 m and the larger deer species (red, sika and fallow) to 1.8 m (Trout and Brunt, 2014). As with browsing, the height of fraying will depend on the species and can be up to 1.8 m for the larger species (red and sika), 1.5 m for fallow deer, 1.2 m for roe deer, and 1 m for muntjac deer (Forest Research, 2024b; Trout and Brunt, 2014).

5.1.6.4 Protection against damage

Tree protection methods identified in the literature for deer are shown in Figure 13, with details and evidence base synthesised below.

MONITORING BEFORE INTERVENTION

Woodland Impact Surveys can be carried out to estimate the current level of deer impact at a site and monitor change over time, with the aim of simplicity of application (The Deer Initiative, 2024). There are alternative methods such as the nearest neighbour damage method that may be more appropriate for commercial crops (The Deer Initiative, 2024). Woodland Impact Surveys look both for signs of deer themselves (seeing deer, dung, racks, slots and other deer field signs) and signs of deer impacts such as fraying, bark stripping, broken stems, browsing (and browse lines) (The Deer Initiative, 2024). They also assess damage to saplings and signs of grazing on the understorey (for example, more bramble tends to mean fewer deer). Overall, this gives a deer activity score ranging from none/minimal to high, which may inform the need for management interventions discussed below. There are also alternative approaches such as the Woodland Herbivore Impact Assessment (WHIA) Method which generates a herbivore impact score (Armstrong et al., 2023) and a WHIA Lite method which is an abbreviated version of the survey (described in NatureScot, 2024). There is a level of subjective assessment in such impact assessment methods which can mean that different surveyors can vary in their impact assessments of the same area. As such, using more than one surveyor and averaging results may provide more robust assessment (Armstrong, 2021).

In a case study of continuous cover forestry in Northamptonshire, Royal Forestry Society (2024b) describe a bespoke impact monitoring method to assess if deer browsing intensity is above thresholds that would inhibit natural regeneration to an extent it would reduce recruitment. This approach uses a small, fenced enclosure and unfenced control area, and relative growth rates between the two help decide if deer control is necessary under current impact levels or if alternative tree protection approaches can be used (Royal Forestry Society, 2024b).

Various deer survey approaches can be used to assess deer density and predict if it is above threshold values that may require intervention, and these include thermal drone surveys (e.g. Hunt, 2024). The density of deer can be used as a risk assessment but may depend on context, for example a lower landscape density of deer may present a risk if areas of woodland are small (for example in farmland) as deer populations may be concentrated in those (Game and Wildlife Conservation Trust, 2024).

REDUCE TREE RESOURCE DEMAND

The aim of culling of deer may be to reduce numbers down to a threshold level below which trees are able to regenerate or colonise new areas in their presence (Forestry Commission, 2021; Hunt, 2024). This target density is context and location dependent and could be as low as 1 per km^2 (e.g. Forest Plastics Working Group, 2024c), and may also differ depending on the availability of alternative food (Woodland Trust, 2022) or fertility of soils (Woodland Trust and Hotchkiss, 2020). What represents a 'tolerable' density (and level of impact) of deer will also likely depend on the function of the woodland, e.g. timber production, conservation, recreation or carbon sequestration (Forestry Commission, 2020b). This reduction in density, but not total exclusion, is argued as an advantage since it allows some deer to be part of the ecosystem while reducing damage risk (Forest Research, 2024b). There is a view that in some other European countries where deer culling is more centrally managed and regulated, deer populations are low enough that need for deer fencing is rare and natural regeneration in woodland is much better (Woodland Trust, 2020). A reduction in numbers may not prevent all browsing regeneration suppression but may be sufficient to allow regeneration of less palatable species such as birch or alder, even if palatable species such as oak and hazel struggle (Forest Plastics Working Group, 2024d).

There is strong evidence that a long-term, large scale deer cull in the absence of fencing can have a measurable positive impact on tree regeneration/colonisation. On Mar Lodge estate in Aberdeenshire red and roe deer were reduced across a >12,000 ha regeneration zone in which a 'zero tolerance' policy was instigated (i.e. all deer detected within that zone were targeted), reducing red deer numbers by about a factor of 10 to <1 per km^2 (Rao, 2017). The number of seedlings with leading shoots browsed within monitoring plots fell from about one third to none and mean seedling height increased (Rao, 2017). Gill (2024) presents a spreadsheet demographic model designed to aid prediction cull effort on future deer numbers.

Deer control is almost exclusively achieved via shooting and current legislation in England and Wales limits most flexibility to undertake any novel control methods (Deer Act 1991, Regulatory Reform (Deer) (England and Wales) Order 2007; David Jam, Forestry Commission, pers. comm.). NatureScot (2024) and The Deer Initiative (2024) provide detailed advice on many aspects of use of firearms and culling of deer. BASC (2024c) have produced a deer stalking code of practice to try to maximise animal welfare and health and safety aspects. Another argued benefit of deer culling via shooting is that this is a recreational activity and a source of income and sustainable meat (Forest Research, 2024c; Hotchkiss et al., 2022; How to Rewild, 2024b). General reduction of deer numbers across the countryside may also rely on a sustainable venison market, and there is some evidence that the infrastructure for such a market is improving (e.g. Ridler, 2024). Venison produced from wild animals may be attractive as a sustainable food source and alternative to farmed meat (How to Rewild, 2024b). Recently, a British Quality Wild Venison Standard has been launched in England, aiming to increase the market for wild venison and protect woodlands (Department for Environment, Food and Rural Affairs et al., 2023). There are also moves towards using lead-free ammunition for deer control to reduce impact on the human food chain (e.g. Woodland Trust, 2020). Deer control is almost exclusively achieved via shooting. NatureScot (2024) provide detailed advice on many aspects of use of firearms and culling of deer, and BASC (2024c) have produced a deer stalking code of practice to try to maximise animal welfare and health and safety aspects

One aspect that may be critical to deer culling as a tree protection method is cooperation with neighbours, particularly if on relatively small landholdings (Forest Research, 2024a), since without this deer may just recolonise areas rapidly. Indeed, it has been suggested that for sites <10 ha deer control is likely to be less effective and other options such as fencing more suitable (Woodland Trust, 2022). On a landscape scale, coordinated deer culling can generate large-scale natural regeneration in the absence of fences, as demonstrated in the Cairngorms Connect project (Gullett et al., 2023). This may be particularly important with the wider-ranging, herding species (Forestry Commission, 2020b). Collaboration can occur between different types of landowners (private, charity, public) and can also involve use of shared facilities such as deer larders (Forestry and Land Scotland, 2023) and can be facilitated via Deer Management Groups (Hotchkiss et al., 2022). A study in East Anglia showed that across four species of deer (muntjac, roe, red and fallow), the effectiveness of deer culling was scale and species dependent (Fattorini et al., 2020). Reduction in impacts from the smaller, more sedentary species (muntjac and roe) was possible with more localised culling efforts while for the larger herding species (fallow and red), reduction in impacts was more effective with a larger scale culling effort. While in the previous study more localised culling was effective in the two smaller deer species, Wäber et al. (2013) did find evidence that forest habitats in East Anglia act as source populations for roe and muntjac deer which would increase chance of recolonisation following culling.

Something critical to deer control within woodland is design aspects of new or restructured woodland to facilitate culling (Cross and Collins, 2017; NatureScot, 2024). This can factor in ensuring open space within the woodland facilitates shooting, such as placement and size of rides and glades: aspects such as aspect, degree of surrounding cover, use of salt licks and placement of hides or high seats can improve ability to control deer (NatureScot, 2024). Apart from at small sites (although what is defined as a 'small site' is not indicated), leaving 15-20% open space for the function of deer control (Forestry Commission, 2020b). These factors should be considered at the planning stage for new woodland, or else gradually introduced over time if managing existing woodland that is lacking these (Cross and Collins, 2017).

Contraceptives administered via injection have been trialled in the USA on white-tailed deer and have been suggested as a potential tool in a European setting where deer numbers are high

(Massei, 2023). However, we found no evidence of this being considered in a UK context, and the logistics are likely to be very challenging even if it were to be effective for individuals.

Diversionary feeding for deer can include provision of plant materials brought in from outside and/or sowing grass, sometimes with added provision of mineral or urea blocks, (Armstrong and Robertson, 2013; NatureScot, 2024) although supplementary feeding of deer can cause aggregations that may increase disease risk (Mysterud et al., 2023). Armstrong and Robertson (2013) also suggest that diversionary food has been enhanced/provided via thinning plantations or grazing cattle to encourage palatable diversionary grass species. Directly supplying food can be labour intensive and can lead to issues such as disease transmission at feed sites, and it is typically best used on a smaller scale top protect small areas of particularly vulnerable or valuable trees (NatureScot, 2024). It is also important to consider that deer may become reliant on the supplementary feeding, such as older animals (NatureScot, 2024). Withdrawal of feeds can lead to mortality or an increase in tree damage to compensate the loss of food (NatureScot, 2024). NatureScot (2024) suggest that the evidence base for diversionary feeding as a tree protection tool is inconclusive, as some studies show reduction in damage, some no change, and some an increase in damage. In some cases, if diversionary feeding is relatively close to young trees it could attract deer to an area causing tree damage or it could, in the long term, increase deer numbers (Armstrong and Robertson, 2013).

MODIFY ACCESS TO TREE RESOURCES BUT NOT AVAILABILITY

The effectiveness of fencing on reducing deer damage and other woodland habitat characteristics has been widely studied using control-intervention study designs (e.g. Game and Wildlife Conservation Trust, 2024). It should be noted that fencing as an effective deer exclusion method is dependent on integrity and maintenance of the fence and a single breach can put a whole planted area at risk (Forest Research, 2024c). Indeed, Palmer (2019) reports a case study of the loss of 40,000 conifers as the result of one hole in a deer fence. Unless maintained/replaced the positive impact of fences on tree regeneration is likely to be relatively short term (see Mitchell et al., 2019). Fences also present a possible risk for bird strikes, such as black grouse in upland areas, and access by the public needs to be managed as well (Forest Plastics Working Group, 2024d, 2024c). Because of the risk of incursion of fences, and because fencing can mean increased deer impacts on the area outside of the fencing, some sources suggest that fencing should be used in conjunction with lethal control to maximise effectiveness (Forestry Commission, 2023; Woodland Trust, 2022). Additionally, fencing can increase deer impacts outside the fence, for example corralling deer along water course sides causing erosion (Cross and Collins, 2017). There is also an argument for fencing several smaller areas rather than one big area due to ease of breaching of fences (Cross and Collins, 2017; Forestry Commission, 2021). Several smaller fenced areas can also facilitate rights of way and reduce cost on gates and stiles, although can increase visual impact and hinder easy access for management (Trout and Brunt, 2014). Deer caught within a fenced area when first established will require culling (Woodland Trust, 2022). Where the fenced areas are very large, thermal drone technology has been used to locate and count deer in order to facilitate control (Forestry and Land Scotland, 2022b). Fencing specification is critical as deer may be able to get through surprisingly small meshes (for example roe deer can squeeze through and A4 paper-sized gap: Yorkshire Dales Millenium Trust, 2019). Fencing height and mesh size can depend on which species are present or being excluded, with generally taller fences needed for larger deer species but mesh sizes not needing to be as large (NatureScot, 2024).

The Deer Initiative (2024) describe temporary fencing as 'providing breathing space' for coppice stools during the initial phase of regeneration. At a site in Nottinghamshire plastic fencing has been used so that it can be moved around the woodland to different sites as necessary (Royal Forestry Society and Forestry Commission, 2022). Cross and Collins (2017) show a type of moveable A-frame deer fencing which is suitable on more uneven ground and can be moved around to promote a diversity of stand structure within native woods. Fencing heights are recommended as 1.8 m for large deer (red, sika or fallow) or smaller deer (roe and muntjac) over larger areas , while for small deer over smaller areas 1.5 m is considered sufficient (Trout and Brunt, 2014). Electric fencing can be used to exclude deer (Forestry Commission, 2023) although there is mixed evidence with suggestions it may not be effective in some cases and is more appropriate for domestic stock (Forest Research, 2024c).

The Deer Initiative (2024) suggest that piling brash around coppice stools or dense dead hedging might be useful for coppice stool protection but are likely to break down after 2-3 seasons. Use of brash that is 'thorny' around coppice stools has also been suggested as a protection tool (How to Rewild, 2024b). Dead hedging built from brash left post-harvest has been used in Oxfordshire as a deterrent for deer browsing for when a compartment is replanted (Royal Forestry Society and Forestry Commission, 2022) although it is not clear if this was effective. In new native woodlands, brash from nonnative trees/shrubs being removed, such as rhododendron or conifers can be used for this purpose (Cross and Collins, 2017).

A related approach to dead hedging is the use of deadwood fencing, proposed by Bradfer-Lawrence and Rao (2012). This is a more structured form of dead hedging that consists of creating a lattice structure several rows high made with long logs (diameter > 10 cm) of felled trees, with planted trees in square gaps within the lattice. To further impede deer access, brash was piled on tip of the logs. Trialled in Scotland, after several years the mean tree height of protected trees was 350 cm compared to c. 150 cm for trees in conventional tree tubes.

As an alternative to the above exclosure methods, in Europe an approach of winter enclosures is used, in which red deer are lured into fenced winter-feeding sites by supplementary food and kept enclosed until spring (Menke et al., 2019). This is said to strongly reduce bark stripping and other forms of browsing over winter, although there are high costs to this approach and other potential issues such as disease transmission and animal stress (Menke et al., 2019).

There has been some assertion that general retention of lying deadwood might reduce deer browsing pressure by creating physical barriers. For example, in Poland, Milne-Rostowska et al. (2020) found a higher occurrence of naturally regenerated rowan saplings where there was more lying dead wood (measured as summed log length in the vicinity). However, Schwegmann et al. (2023) found no relationship between volume of lying deadwood and roe deer browsing pressure in southern Germany. This this approach requires more investigation across more scales, ranges of deadwood volume and species. Retention of windblown trees has also been suggested as a barrier to deer browsing that can promote regeneration (The Woodland Trust, 2019). In an experimental field trial in Germany, the retention of lying tree crowns reduced the probability of roe deer browsing on planted silver fir saplings (Hagge et al., 2019). In a review of deer browsing impacts across Europe, Gerhardt et al. (2013) identified mix finding with relation to dying deadwood, identifying studies that had found positive effects on regeneration but also negative effects. Armstrong and Robertson (2013) also reviewed several global studies that had investigated lying dead wood such as logs, brash or general woody debris. They also found mixed results, with positive associations with tree growth/survival and negative associations with deer damage in some, but not all, cases.

NatureScot (2024) make a distinction between tree shelters (often called tree tubes) which have solid sides and tree guards which are plastic or wire mesh, the latter of which protect the bark and terminal shoot but allow lateral shoots to grow through the guard and may have a lower visual impact if this is important. Nevertheless, in terms of height, for larger deer (red, sika and fallow) a height of 1.8 m is recommended, and 1.2 for roe deer muntjac deer (NatureScot, 2024; Trout and Brunt, 2014). It should be noted that on steeper slopes the effective height of a tree tube is reduced as deer and other grazers can use the higher side to reach above the tube level (Yorkshire Dales Millenium Trust, 2019). Tree tubes may be less suitable for conifers (NatureScot, 2024). It should be noted that tree tubes create a warmer and more humid microclimate that appears to benefit some species but may be detrimental to others (Forestry Commission, 2020a). Thus, choice of solid tube versus open mesh guard may depend on tree species . The effectiveness of tree tubes/guards may depend on their placement, and they can be knocked over by deer, particularly if not robustly staked in, and the cable ties can be nibbled by deer (Yorkshire Dales Millenium Trust, 2019). Tree tubes of a specific design have been shown to be effective at substantially reducing fraying damage by roe deer to newly planted orchard fruit trees in the Czech Republic (Marada et al., 2019). Furthermore, this was found to be even more effective if 'rendering fat' was applied to the tube, although the mechanism was not described.

The choice of using tree tubes/guards versus fencing can depend on the area being protected (Cross and Collins, 2017). Typically, the cost per hectare for tree tubes remains the same or similar no matter how large the area, while for fencing the cost per hectare declines rapidly as the area protected increases (NatureScot, 2024). This means there is a threshold area below which tree tubes/guards may be more economical and above which fencing may be more economical (NatureScot, 2024). Forestry Commission (2020a) suggest that where deer are present, tree tubes or guards are most suitable where new woodland is smaller (< 3 ha), non-commercial, and where deer control is absent or has not reduced damage to an acceptable level. Fencing cost is also increased by the considerations of amenity/access, the need to be robust to vandalism, and visual impact , the latter which may increase length required if straight lines are avoided (Trout and Brunt, 2014). Tree tubes/guards can allow positive herbivore impacts on ground vegetation while protecting trees themselves, although if herbivore vegetation impacts in an area are generally negative then then this benefit becomes a risk (Trout and Brunt, 2014)

Chau et al. (2021a) report on a lifecycle analysis of different tree tube options, which looks at the carbon costs of different options, including using no tree tubes/guards and suggest that carbon stored in a tree over its first 25 years of growth outweighs that emitted via manufacture, transport and decay of plastic tree tubes. However, the authors also found that the balance of carbon costs depending on rate of sapling survival in the absence of tree tubes. If relatively high, then the carbon costs of replacing these via periodic planting is less than the carbon cost of using tree tubes, whereas if sapling survival is low (below a third or so) then the carbon cost is lowest if plastic tree shelter are used. Chau et al. (2021a) also found that the carbon cost of non-plastic tree tube alternatives made from poly-lactic acid or bio-polypropylene resins was higher than those of normal plastic tree tubes because of much higher carbon costs in manufacture. There are also some examples of non-plastic tree tubes being bitten through by deer, tearing, or else sagging/slumping (to the extent they can squash the sapling inside) (Yorkshire Dales Millenium Trust, 2019).

Within parklands or wood pasture, parkland tree guards are often used which have many different designs, but often consist of wooden frames with mesh (although can be made from ironwork) and are essentially small enclosures around individual trees (How to Rewild, 2024b; NatureScot, 2024).

In terms of artificial tree protection approaches, even more localised than tree tubes or guards is the use of individual bud protectors. These have been made from paper, foil, netting, or specially designed plastic bud caps and can be placed on the leader shoot (Armstrong and Robertson, 2013). These are intended to move (or be moved in some designs) with the bud as it grows while deterring browsing of the leader specifically. Mesh leading bud protectors have been designed that naturally degrade, allowing the bud to burst out the following growing season from when it is protected (Armstrong and Robertson, 2013). These approaches have yet to be widely tested in the UK to our knowledge.

Seeding oak directly into clusters of planted thorn trees is a method adopting the associational resistance effect which has been used in a woodland creation scheme in Wales (Burton, 2024). In management guidelines for native woodlands in Ireland, Cross and Collins (2017) also suggest retaining bramble for this reason. However, they also recommend cutting any bramble shoots that are crossing over young trees to aid crown release. Armstrong and Robertson (2013) carried out a review of various studies that had investigated associational resistance against deer and livestock. Protective thorny/spiny species in the UK featuring in the review included raspberry, bramble, holly, blackthorn and hawthorn and generally studies found a positive effect on protected tree growth or survival

NatureScot (2024) and Trout and Brunt (2014) suggest that chemical repellents may be best used as an ‘emergency option’ if immediate protection may be required. Their high cost and generally limited duration of effect means they are likely to be more appropriate to protect small areas of particularly valuable or vulnerable trees. Because the UK has a stringent regulatory/testing regime for repellents combined with a relatively small market, a limited number of repellents are approved by the Chemicals Regulation Directorate (CRD) in the UK (currently to our knowledge: Curb Crop Spray powder, Liquid Curb Crop Spray, Rezist and Sphere ASBO, all of which are based on aluminium ammonium sulphate). However, it is feasible to receive a license to trial new method over a small area (Armstrong and Robertson, 2013).

Chemical repellents for trees can be systemic (i.e. they are taken up by the plant) or surface based (they are applied to the plant) (Armstrong and Robertson, 2013). The chemical repellent Trico is a smell and taste-based deterrent made from emulsified sheep fat (Palmer, 2019). It has been used to protect hazel coppice stools and although some users report its relative expense, it claims effectiveness for six months (reported by Wildlife Trust for Beds, Cambs and Northants, 2024). It requires manual spraying twice per year and it has been suggested it is a better option for protecting small groups of planting where fencing would be impractical or expensive for such a small area (Royal Forestry Society, 2024c). It has also been cited as a replacement for tree tubes/guards although without reported results (Royal Forestry Society and Forestry Commission, 2022). Anecdotal evidence suggests that Trico treatment to just the perimeter of a stand (to a depth of 3-4 trees) may also be effective (Palmer, 2019). Additionally, foresters have trialled treating transplants before planting and treating the tops of 60 cm tree tubes with Trico to protect growth just emerging from the top (Palmer, 2019). Although costs will be highly context dependent, there is some suggestion that Trico treatment can be a less expensive alternative to fencing on restock sites below 40-80 ha, although if a grant for fencing is available this will change the balance (Palmer, 2019). A chemical repellent based on a plant nutrient calcium chloride called 'Grazers' is available which is apparently widely used in conifer nurseries and Christmas tree farms and according to Armstrong and Robertson (2013) does not need approval by the Chemicals Regulation Directorate (CRD).

Hydrolysed casein has also been tested as a chemical deer repellent in the USA although it has some challenges in terms of application (see How to Rewild, 2024b). Another type of repellent suggested is that based on lanolin (a wax secreted by sheep). This has been suggested as potentially useful at low deer densities, if applied to the most palatable tree species and combined with lethal control (Forestry Commission, 2023). Mrnka et al. (2021) also indicate that where browsing resistant varieties of trees can be identified, and the compounds they contain that are responsible for that resistance, isolation of such compounds could form future plant-based chemical repellents. Some repellents such as capsaicin act as irritants which deters deer (Armstrong and Robertson, 2013).

Other types of sensory repellent tested in the past include have included soap, human hair and urine of humans or deer predators such as wolf (How to Rewild, 2024b). While there is evidence that scent repellents don't work as well as taste repellents (Armstrong and Robertson, 2013), McCallin (2017) tested the effect of three odour-based repellents – rotten eggs, pig's blood and wolf faeces – on red deer space use and behaviour within captive populations. The pig's blood appeared to increase alert behaviour at one site, while all three repellents appeared to lead to some degree of avoidance.

Inclusion of sheep's wool into mulching is considered to have deer repellent benefits (Burton, 2024), although attachment of sheep's wool to saplings has been observed to not deter deer (Yorkshire Dales Millenium Trust, 2019).

Research has shown that deer distribution in a landscape can be driven by a range of factors including food, thermal properties and fear (Widén, 2023), and some approaches aim to use fear as a deterrent. A study in mixed forests in Poland showed that there were fewer red and roe deer closer to forest roads and also less browsing damage or bark stripping (Mathisen et al., 2018). This suggests that fear can reduce deer impact in the absence of direct control or exclusion. Use of acoustic deterrents including dog barking, deer alarm calls or a pure tone have been trialled and shown success with deer, although in contexts such as keeping them away from railway lines or agriculture (discussed in How to Rewild, 2024b; Laguna et al., 2022). There is also some evidence that presence of dogs themselves can deter deer (discussed in How to Rewild, 2024b).

MODIFY TREE RESOURCE AVAILABILITY OR RESILIENCE TO DAMAGE

Where tree protection approaches that do not involve fences or physical barriers, there is an argument that they are likely to work best when deer densities have already been reduced by control methods (Armstrong and Robertson, 2013). Planting of less palatable tree species has been used as one of several deer (and other browser) impacts in a new woodland scheme in Wales, although the impacts are not yet reported (Burton, 2024). Italian alder has also been suggested as a relatively unpalatable species for deer and in a scheme in Oxfordshire has been planted around the edges of a conifer block although not tested for effectiveness (Royal Forestry Society and Forestry Commission, 2022). In some cases, specific browsing resistant varieties of tree species have been tested. For example, Mrnka et al. (2021) show in a feeding trial with captive red deer that some varieties of hybrid poplar *Populus nigra* × *P. maximowiczii*, a tree used in short rotation coppice in the Czech Republic, were less attractive to deer due to certain compounds they contained. The authors also suggested these compounds could be extracted and tested as a chemical repellent for use on other species.

Because of the attraction of certain softwood tree species for bark browsing, such as willow, rowan and aspen, it has been suggested retaining these species as a form of sacrificial planting (Konopka et al., 2022). The authors also suggest actively planting small (e.g. 0.25 ha) browse plots of these species among commercial stands, although these were not tested (Konopka et al., 2022). Sacrificial planting for fraying, such as use of willow on forestry compartment boundaries has been suggested (NatureScot, 2024). Similarly, it has also been suggested in some cases that overstocking density of trees (such that some level of damage is accepted with the hope a proportion will survive) may be an approach to reach target stocking densities without other tree protection methods (Woodland Trust, 2022).

Another approach that has been suggested is clump planting whereby trees are planted in dense clumps, relatively cheap and fast growing species such as willow or birch can be planted on the outside to act as a sacrificial feed and/or a physical barrier to access to the desired planted trees in the centre (Armstrong and Robertson, 2013). The desired tree must have sufficient growth rate to not be swamped by the protecting trees, and this approach had not been trialled in the UK when discussed by Armstrong and Robertson (2013).

A study by Bolibok et al. (2021) into planting pattern and vegetation management of oak plantations in Poland found that the choice of planting and management could reduce browsing pressure by deer in the absence of fences. Browsing damage was lowest where group planting was used (oaks are planted in groups of 24 with approximately 5 m gaps between groups) and competing naturally regenerating species that grew within the gaps were topped (i.e. cut at the level were leaders were removed) in spring. In contrast, more traditional planting patterns or double row planting with traditional ground-level removal of competing species resulted in significantly higher browsing damage to oaks.

Sabre planting, which involves planting taller (≥ 1.2 m tall) saplings perpendicular to steep slopes (at least 30-40°) so that they are out of reach of browsers (livestock and deer) has been suggested (Armstrong and Robertson, 2013; Woodland Trust, 2022), although this would not be suitable for timber crops. This approach has been tested with ash, sessile oak, downy birch and alder is reported to have been effective against roe deer browsing but has not been tested with red deer (Armstrong and Robertson, 2013). The leading shoot and the shoots on the downslope side are less accessible to browsing animals . In upland areas, montane willows have been planted on inaccessible ledges to protect them from red deer browsing.

For species such as willow which can grow from cut rods, planting willow pegs below ground level so that roots can establish before they sprout (Woodland Trust, 2022). Additionally, 3 m willow forks whereby they are pushed into the ground far enough to prevent pulling out but have buds above browsing heights, have been suggested as approaches to encourage willow regeneration in the presence of browsers such as deer and others (Woodland Trust, 2022).

An experimental study in Oxfordshire showed that increased coppice height substantially reduced hazel coppice stool browsing and mortality (Wright and Bartel, 2017). Roe deer and muntjac deer were known to be present, and stool mortality decreased from c 96% for ground level stools to 0-8% for stools 0.7-1.2 m high. The mechanism suggested is that at 0.7 m, the leader shoots are above (or shortly after cutting will grow above) the browsing level of the smaller deer species.

Management of cover has been suggested as potential tree protection approach for deer, as some species such as roe deer are associated with the availability of hiding cover (Schwegmann et al., 2023). Abundance of cover was suggested as a risk factor for deer browsing impact in a review by Gerhardt et al (2013), which is likely due both to thermal cover and anti-predator behaviour.

In a review of deer browsing impacts across Europe, Gerhardt et al. (2013) identified clearcutting regimes were most vulnerable to deer browsing impacts and that shelterwood felling, or group or single tree selection felling can reduce this impact.

As discussed, one of the sublethal effects of deer (and other species) damage can be rot formed from bark-stripping/fraying wounds, and one such rot can come from infection by *Heterobasidion annosum* which creates “butt rot” in Sitka spruce and can reduce timber value significantly. A study by Pratt and Thomsen (2022) investigated the possible use of the biological control agent *Phlebiopsis gigantea* to prophylactically reduce butt rot in species such as Sitka spruce. However, this work is still at an exploratory stage, and it appears that the level of infection of butt rot in the UK is relatively low relative to other parts of Europe.

5.1.7 Livestock: Sheep, goats, cattle, and bison

5.1.7.1 Background and ecosystem services

The presence of large herbivores in woodlands, guided by woodland management objectives, can maintain the conservation interest of grassland areas in woodland mosaics (Scottish Forestry, 2024; Woodland Trust, 2022). This can maintain open areas within woodland (creating habitats for ground flora and invertebrates) and may help to preserve archaeological remains (Scottish Forestry, 2024; Woodland Trust, 2022). Silvopastoral systems are predicted to store more carbon than equivalent areas of separate woodland and pasture, although the differences are small and may not be detectable empirically (Upson et al., 2016).

Maintenance of open areas may benefit wood ants, woodland specialist butterflies and spiders while animal dung provides a resource for beetles (particularly *Geotrupes*), many species of flies, earthworms, nematodes, mites and springtails. This can attract in turn jackdaws, waders, chough, starlings, woodland grouse, badgers, foxes, shrews, hedgehogs and bats. Low to moderate woodland grazing will favour wood-warblers, pied flycatchers, redstarts, thrushes and tree pipits which need low shrub cover. Trampling can be a risk to ground nesting birds but can be mitigated by introducing stock after birds have fledged. Heavily grazed woodlands have reduced populations of small mammals with consequent impacts on their predators (Mayle, 1999).

In the pre-Neolithic period (c. 7000 years ago), large herbivores could have been a significant factor in shifting the balance of woodlands towards oak and away from more shade-tolerant species (Hodder et al., 2005). Bison, auroch and horses were present in western Europe before settled agriculture, but sheep and goats were Neolithic introductions (Small, 2004). That bison are native to Britain is plausible, although unproven (How to Rewild, 2024c).

Sheep typically flock in groups of about 100 animals occupying a home range bounded by topographic features such as ridges or watercourses. Subgroups of related ewes are hefted to a smaller home range – usually where they were reared - which includes a range of vegetation types. Sheep tend to spread out while foraging, to move uphill at night and in summer, and to seek the shelter of walls or woodland in winter (Scottish Forestry, 2024). Sheep are highly selective grazers, eating more live vegetation, fine-leaved grasses and tender shoots but less fibrous material and creating a short, tight sward (Woodland Trust, 2012a). Sheep are considered to have similar browse preferences to red deer, favouring firstly ash, aspen, willow, hazel, holly and juniper;

secondly oak, thirdly birch and pine, and fourthly alder and juniper. Sheep are thought to selectively browse trees growing on more nutrient-rich soils (Andrews et al., 2000).

Cattle move around their home range as a herd, congregating around a water source in hot weather. They are more likely to spread out across their range if they spend much of the year on a site and become more familiar with it. Being larger, they cause more trampling and poaching impact than sheep and may be more effective at reducing bracken cover (Scottish Forestry, 2024). Cattle are not particularly selective in their grazing, and will eat dead herbage, sedges, rushes and other tall, fibrous elements of the sward not palatable to other grazers, creating an uneven, tussocky sward. (Woodland Trust, 2012a). Trampling by cattle can be beneficial to create areas for light-demanding trees such as oak to regenerate, a function absent in many woodlands because wild deer do not trample to this extent (How to Rewild, 2024d)

Wild water buffalo *Bubalus arnee* are native to south and southeast Asia (IUCN, 2016) but animals of a domestic breed *B. bubalus* have been used for conservation grazing in some UK wetland reserves (e.g. Gulickx et al., 2007). Similarly, European bison which are found in fragmented populations in eastern Europe (IUCN, 2020) and have been introduced into ancient woodland in Kent where their bark stripping behaviour is expected to provide deadwood, while dust bathing providing habitats for pioneer species (https://www.rewildingbritain.org.uk/rewilding-projects/wilder-blean). Bison range over 100km^2 or more, favouring open areas and grass but also eating leaves and twigs. Different suites of plant species are dispersed in bison dung, hooves and fur (How to Rewild, 2024c).

Goats may not make useful conservation grazers because they are considered less selective in their diet, can readily escape, are non-native and not functionally analogous to absent native herbivores (How to Rewild, 2024e) (https://www.lrwt.org.uk/blog/fran-payne/conservation-grazing-what-it-and-why-do-we-do-it). Goats are random browsers, moving regularly, although they may return to areas they know to provide good forage, escaping enclosures to do so (How to Rewild, 2024e). Goats tend to eat above ground level and favour high quality grasses. They can control bramble and other thorny species but browse broadleaf seedlings more than other livestock, preventing natural regeneration. They create uneven, tussocky swards (Woodland Trust, 2012a). Feral goats are restricted to areas where they have access to shelter from rain (Scottish Forestry, 2024). Goats can affect regeneration on cliff ledges or boulder fields inaccessible to other grazers (Andrews et al., 2000).

Autumn is the best season for livestock grazing in woodland because biomass will be at its maximum, if not grazed in summer. Browsing damage to woody species is likely to be greater in winter when alternative forage is less available. Wet soils in winter are most vulnerable to poaching but lesser degrees of disturbance may help to reduce dominance of bracken and create niches for germination of tree seeds in the following seasons. Heavy spring and summer grazing reduces diversity of the field layer and reduces nectar and pollen resources available from flowering plants. If autumn-only grazing is impractical, low stocking densities and supplementary feed in winter may reduce the impact of year-round grazing (Scottish Forestry, 2024).

The Woodland Grazing Toolbox (Scottish Forestry, 2024) provides guidance on designing a grazing regime to achieve woodland management objective. These, include dividing a woodland into grazing management units (areas which can be appropriately grazed under the same regime), selection of species, breed and grazing season, calculating stocking density, integrating with wild herbivore management and monitoring the effects of grazing.

Livestock, as analogues for extinct wild herbivores, form part of trophic rewilding schemes where they influence the dynamics of habitat mosaics including early-successional woodlands. Tanentzap et al. (2023) studied the effect of old English longhorn cattle (at densities between 18-60 km-^{2}, usually 20-30 km^{-2}), Exmoor ponies (3-5 km^{-2}) and Tamworth pigs (2-9 km^{-2}) together with fallow (7-55 km^{-2}), red (0-7 km^{-2}) and wild roe deer over nine years at Knepp Estate in Sussex. Total livestock units per hectare ranged from 0.13 to 0.37 during this period. They found that woody plant diversity was reduced by 73%, carbon storage (woody, herbaceous and below-ground) reduced by 23% and vegetation height restricted by large herbivores to about one-fifth of the height compared to plots from which they were excluded. However, structural complexity was greater when herbivores were present, resulting in 21% greater diversity of ground-dwelling arthropod families and 167% greater arthropod biomass.

Wood pasture is a mosaic habitat valued for individual trees, particularly veteran and ancient, and the fauna, flora and fungi it supports, including a number of species that only occur in wood pasture and parkland. Dung contributes to invertebrate and fungal diversity and grazing controls young trees and shrubs, maintaining the semi-open habitat, although over-grazing risks damaging floral diversity and preventing the regeneration which allows for eventual replacement of today’s veteran trees (Woodland Trust, 2012b).

Livestock grazing, by reducing the competition of weeds with young trees, may reduce herbicide requirements in some circumstances (Mayle, 1999). Grazing of sheep in apple orchards reduces the need for mowing and is proposed by stakeholders to offer a possible means of controlling apple scab while minimising pesticide inputs because sheep eat and trample fallen apple leaves, reducing harbourage for the organism responsible (Pantera et al., 2018).

Armstrong et al. (2003) surveyed site managers at 105 sites in Britain where cattle are grazed in woodlands. The majority (85%) of cattle-grazed woodlands in Britain are semi-natural and most (69%) are dominated by either oak or birch – in England, particularly oak. Cattle-grazed woodlands vary greatly in size, but most are less than 50 ha and almost all are grazed by other herbivores in addition. Nature conservation and, secondarily, production were the main motivations for cattle-grazing woodland and sites did not differ in grazing intensity according to aims. Conservation-related aims included encouraging tree regeneration (including oak, birch, aspen and Scots pine), benefitting individual taxon groups, preventing tree/scrub regeneration, maintaining open areas, reducing shrub layer (particularly bramble) or reducing dominant plant species in the ground layer. In cattle production, obtaining shelter from the woodland was at least as important as forage. Nearly two-thirds of sites were grazed in summer although there was much variation in the combinations of grazing seasons. 31% of sites had stocking densities at or below 0.1 cattle/ha/year

(1.2 cattle-months/ha/year) which is commonly recommended for conservation grazing and only five sites had grazing intensity greater than 10 cattle-months/ha/year. The probability of finding no regeneration increases with stocking density. The order of preference of saplings for browse was oak, goat willow, Scots pine, birch, ash, hazel, beech, alder, rowan, holly, hawthorn (Armstrong, Poulsom, Connolly, & Peace, 2003).

Grazing of sheep, cattle, horses and/or goats increased the plant species richness of broadleaved woodland in Northern Ireland when examining small plots (14 versus 10.5 species per $4m^2$ plot) but did not have a detectable effect when examining larger ($196m^2$) plots. Cover of bramble *Rubus fruticosus* (3.9% versus 9.6%) and bluebell *Hyacinthoides non-scripta* (2.5% versus 10.1%) were significantly reduced by grazing (Agra et al., 2016).

Cattle grazing at 59-104 animal-days per hectare for two months in the autumn at one, unreplicated pinewood site (NVC W18b community) in the Scottish Highlands led to vegetation changes likely to favour Scots pine recruitment (reduced moss/litter depth and increased ground-level light incidence). However, seed fall was insufficient to detect any effect on recruitment. Cattle increased bilberry *Vaccinium myrtillus* cover by 1.9 times compared to control, directly related to trampling impacts on heather (Hancock et al., 2010).

Any planned introduction of grazing requires knowledge of animal husbandry beyond the scope of this review. Fencing, a water source, vehicular access and handling facilities will be required, as will regular checking of animal welfare and perhaps agreements with other landowners if livestock are to be moved seasonally (Woodland Trust, 2022).

5.1.7.2 Damage and impacts

The most frequent concern regarding overgrazing in woodlands is lack of tree regeneration. Sheep, in particular, prevent recruitment by eating seedlings and young trees (How to Rewild, 2024f). Regrowth from coppice stools may be reduced by livestock and established trees damaged by bark stripping (Thompson et al., 2005). Establishment, thicket and pole-stage trees are most likely to be damaged by livestock browsing, although newly planted trees may also be pulled out by ponies, cattle or sheep (Forest Research, 2024b).

Fifty-eight herders who grazed livestock in forests in Hungary most often discussed impacts of sheep on leaves of woody species, shrub layer, herb layer, wild fruits, bark, acorns and timber trees (in descending order of frequency). For cattle, the most discussed impact was on the herb layer, followed by shrub layer, leaves of woody species, soil quality (including fertilisation), twigs, grass density and timber trees (Varga et al., 2020).

Sheep are sometimes considered to be incompatible with tree regeneration because of browsing on seedlings and saplings (Woodland Trust, 2012a). They will also strip bark, especially in harsh winters. Sheep grazing among oak in the Forest of Dean at sufficient intensity to prevent regeneration, leading to high-forest with even-aged trees and little shrub layer, reduced the diversity of winter bird communities and led to the absence of indicator species of ungrazed oakwoods such as hawfinch (Hill et al., 1991).

Trampling and poaching can be negative or can create niches for rare plants (Spencer, 2023) and bare patches for ground-nesting species (Mayle, 1999). Grazing may accentuate the competitive advantage of tufted hair grass *Deschampsia cespitosa* on wet ground, but lack of grazing may allow competitive woodland ground flora such as wood anemone *Anemone nemorosa* and bluebell *Hyacinthoides non-scripta* at the expense of smaller plants (Mayle, 1999). Cattle may break up dense stands of bracken which can smother tree seedlings (Armstrong, 2013) and have a greater poaching effect than other livestock (Woodland Trust, 2012a). Sheep are considered to compact the ground more than cattle (Woodland Trust, 2012a). Fenced oak-birch woods have higher densities of fine roots and mycorrhizae than adjacent sheep-grazed woods (Pigott, 1983).

Šumavka sheep at 3.75 livestock units/ha in two-week intervals from June to September in sessile oak *Quercus petraea* and hornbeam *Carpinus betulus* coppice increased the number of resprouts per stump, reduced the height of resprouting and decreased the diameter of sprouts (although the latter only in combination with litter-raking). Sheep grazing and litter raking in combination decreased the circumference gain of oak standards (Kadavy et al., 2019).

Exclosures were established in 1955 and in the 1980s to study the effect of sheep grazing in Yarncliff Wood, an ancient oak-birch wood in the Peak District National Park which had been intensively grazed (from 0.7 sheep ha^{-1} in the 1930s to 2.07 sheep ha^{-1} in the 1970s). Areas still grazed in 2011 had no regeneration of woody species, no shrub layer and limited ground flora. In the 1980s exclosure, shrub cover was 8%. In the 1955 exclosure, 15% cover of shrubs had developed and beech and holly had colonised, which were absent prior to exclosure. There was declining dominance of light-demanding oak and birch with increasing presence of shade-tolerant rowan and beech. Although initially change in the 1955 exclosure had been rapid, little change in the ground flora was observed from the 1980s to 2011 except for the decline in cover of previously co-dominant bracken (Vild and Rotherham, 2021).

Tree and sapling communities in upland Atlantic oakwoods in Wales intensively grazed by sheep or feral goats for at least 25 years had lower seedling establishment, sapling recruitment and functional diversity than equivalent ungrazed (or lightly winter-grazed) woodlands. Plant traits and functional diversity did not differ with grazing intensity for understorey plants. However, tree and sapling communities from grazed woodlands were characterised by grazing avoidance strategies (high leaf dry matter content and low specific leaf area i.e. tougher leaves), higher shade-tolerance and higher drought-tolerance but lower water-logging tolerance than in un-grazed woodlands (Ford et al., 2018).

Pollock et al. (2005) studied sheep grazing with or without cattle (2.7-166.8 LSU d ha^{-1}) and with wild deer/hares in seven upland birchwoods in Scotland, finding birch regeneration compatible with livestock grazing. Within-site variation in the proportion of shoots browsed per sapling was high, but browsing intensity was lower when good quality biomass per livestock unit was high. Trees with greater basal diameter were browsed less and trees with taller adjacent vegetation were browsed less. Saplings with a topiaried growth form after repeatedly losing their leader were browsed more than saplings with a normal growth form. Pollock et al. (2005) predict that more than 10 kg per LSU day of good quality biomass is required where vegetation adjacent to trees is short.

Exclosure of cattle from an Irish ancient lowland wet oakwood for ten years led to increases in regenerating ash *Fraxinus excelsior* and holly *Ilex aquifolium* compared to light grazing, likely over time to shift dominance from oak to ash. There was also a decrease in the cover of ruderal plants and grasses and some establishment of non-native trees. Changes were similar under low and high canopies but more marked under high canopies (Cooper and McCann, 2011).

Palmer et al. (2004) studied Atlantic oakwoods at five sites in Western Scotland grazed by sheep, roe and red deer at intensities ranging from <10 to >120 animal days/ha. Incidence of browsing varied between sites and years with a summer minimum of 20% and a winter maximum over 70%. Rowan and birch were more browsed than oak and hazel in summer, but no tree species preference was evident in winter. Even unbrowsed oak saplings failed to grow, probably due to shading, and the authors considered that ungulate browsing was unlikely to be the significant factor limiting regeneration.

Goats preferentially select scrub and tree leaves, climbing trees or pulling branches through fences to reach them, preventing tree regeneration and preventing or reversing scrub encroachment of open areas. They strip bark from deciduous trees and young trees are often ringbarked (How to Rewild, 2024e; Log et al., 2022). Goats will browse scrub and hedgerows by climbing among the stems, creating gaps which may allow other stock to escape (How to Rewild, 2024e).

Bison rarely uproot or strip bark from mature trees but will graze tree saplings, maintaining open areas (How to Rewild, 2024c). Reports of bark stripping vary from 1.4% of all trees damaged by bison (How to Rewild, 2024c) to 20% of available trees (Broughton et al., 2022) perhaps due to low availability of preferred foods. As for other large herbivores, damage by bark stripping or horn rubbing may result in rot (80% of bark-stripped beeches in a German bison reserve, Kelterborn 2009 cited in How to Rewild, 2024d) which might create cavities used by birds (Broughton et al., 2022). Bobiec *et al.* (2011, cited in (Armstrong and Robertson, 2013) found that oaks regenerated in Białowieża National Park in Poland in the presence of mixed large herbivores including bison, deer and boar although Smit *et al.* (2012, cited in (Armstrong and Robertson, 2013) concluded that large woody debris in this forest was important to protect young oaks against this mix of ungulates.

Murphy et al. (2022) studied colonisation of *Quercus* in pastures adjacent to upland oakwoods in Dartmoor National Park. Winter sheep (0.012 LSU ha−1) and summer cattle (0.201 LSU ha−1) grazing removed all of the 8–12-year-old oak saplings which were present at an average density of 200 per hectare within exclosures. However, there was no significant effect on density of younger saplings, in fact, sites with youngest saplings (1–3 years) were dominated by open, grazed swards, indicating that grazing facilitates the initial stage of colonisation even though the same intensity of grazing prevents succession to woodland. Sapling height gain was significantly greater where livestock were excluded. At nearby sites, saplings in extensive pasture (grazed by sheep, cattle, ponies at 0.400 LSU ha^{-1} in summer, 0.170 LSU ha^{-1} in winter) had lesser, although more variable, browse damage and better growth than saplings in enclosed pasture (Murphy et al., 2022).

Juniper *Juniperus communis*, a native shrub of conservation interest, is browsed by cattle and sheep as well as horses, deer, rabbits and rodents. Although juniper is unpalatable and has even been associated with deaths of livestock, horses have been recorded to kill juniper by gnawing the bark. Other browsers can significantly alter the shape of the bushes and large animals including cattle can fragment dense stands, even eliminating juniper in extreme cases. Browsing of juniper is more likely in winter and when other forage is scarce. Grazing of seedlings limits juniper regeneration but some grazing may be required to open up dense vegetation to allow germination. Juniper has also been associated with sheep walks, which are said to disperse its seeds (Thomas et al., 2007). In an unreplicated study in southern Sweden, juniper seedlings were more than twice as abundant in cattle-grazed limestone heath as in a comparable ungrazed plot, although seedlings in the grazed area appeared more vulnerable to drought (Rosen, 1988 cited in Thomas *et al.,* 2007).

Goats frequently browse on thorny shrubs and sometimes on chemically defended shrubs if those form a substantial part of the vegetation. Elias and Tischew (2016) observed that the effect of goats on abandoned pastures in central Germany at 0.6-0.8 LSU/ha/year (6-8 goats per hectare from spring to autumn). Goats reduced the coverage of thorny shrubs including *Berberis, Crataegus, Prunus, Robinia* and *Rosa* from 70% to 37% over seven years, converting scrub to grassland, despite the opposite trend on ungrazed plots. Goats spent more time browsing in spring than summer or autumn which may translate to a greater impact on vegetation. Most species were browsed in proportion to their abundance but invasive *Robinia pseudoacacia* was preferentially selected.

Forage intake can be estimated from the weight of the animal and the digestibility of available forage. The Woodland Grazing Toolbox (Scottish Forestry, 2024) gives estimated intakes for a range of species and breeds using the following equation, derived for sheep and applied as a crude approximation for other species (Equation 1):

Intake (kg. dry matter/day) = metabolic live weight (kg.) x (0.167 x digestibility) - 0.044 (Equation 1)

where metabolic live weight is live weight to the power of 0.75; summer diet digestibility is assumed to be 0.7, winter digestibility 0.5. For equines which are less affected by forage quality, the summer value can be applied year-round, but the result multiplied by 1.84 to account for the fact that equines consume more than ruminants because of their less-efficient digestion.

Subtracting any supplementary feed, the remaining intake can be assumed to come from woodland vegetation, distributed according to the animals' forage preferences and the available vegetation (Scottish Forestry, 2024).

Based on a Finnish study (Takala et al., 2015), allowing cattle to move between fertilised grassland and forest pasture is not recommended because cattle transport nutrients to the forest in dung and urine which adversely influence the conservation value of plant communities. Dung may be concentrated in patches where animals lie up at night and cattle dung in particular may smother plants. Most livestock avoid grazing near dung patches, allowing rank vegetation to develop (Mayle, 1999)

5.1.7.3 Identification of damage

Sheep and goats will forage on grass, forbs, heather, shrubs and trees, leaving a ragged edge to browsed shoots and eating the removed portion, producing a browsing pattern which may be indistinguishable from that of deer except that wool is likely to evidence the presence of sheep (Forest Research, 2024b; Scottish Forestry, 2024). Newly planted trees may be uprooted. Bark stripping on young to pole stage trees may be severe, particularly from goats, and leaves diagonal incisor marks up to their 1.5m reach (Scottish Forestry, 2024; Thompson et al., 2005).

Cattle may rub against and strip the bark of all ages of trees. They can reach up to around 2m and leave a ragged edge to browsed shoots (Scottish Forestry, 2024). Cattle browse up to 2m, while sheep and goats can reach up to 1.5m (Andrews et al., 2000) or goats to 2m by standing on their hind legs (Elias and Tischew, 2016).

The root systems and mycorrhizal communities of ancient and veteran trees (AVTs) can be damaged by compaction or poaching due to trampling as well as from excessive nutrient input. Microorganisms and soil animals around AVTs can be harmed by the excretion products of veterinary medicines (Woodland Trust, 2021a). Bark-stripping by livestock at high densities is a concern; however, light grazing prevents the buildup of vegetation which poses a fire risk and may threaten lichens or invertebrates by shading the trunk (Woodland Trust, 2005).

5.1.7.4 Protection against damage

Tree protection methods identified in the literature for livestock and bison shown in Figure 14, with details and evidence base synthesised below.

MONITORING BEFORE INTERVENTION

Woodland Herbivore Impact Assessment (Armstrong et al., 2023) can be used to assess the combined impacts of wild herbivores and livestock and, if surveys are timed before and after a seasonal grazing period, the relative contribution of wild and domestic herbivores can be interpreted (Scottish Forestry, 2024).

English Nature (Thompson et al., 2005) developed a method to identify overgrazing in English upland native woodland based on characteristics of the browse line, frequency of seedlings and saplings and degree of browsing on these, bark damage, broken stems and characteristics of the ground flora. The survey is best carried out in winter (Feb-March) and severely affected woods followed up in spring to look for signs of recovery.

In Hungary, herders go into forests with their animals and assess when they can return to an area based on the state of the vegetation and their traditional knowledge (Varga et al., 2020).

REDUCE TREE RESOURCE DEMAND

Lethal control options are available for feral sheep and goats (as well as exotic feral species such as wallabies) and these are governed by the Wild Mammals (Protection) Act 1996 (David Jam, Forestry Commission, pers. comm.).

The Woodland Trust (Woodland Trust, 2022) recommend a period of five to ten years to establish trees before introducing livestock, unless an extended early-successional stage is desirable. If natural colonisation is desired, exclusion of sheep is recommended. The benefit of cattle in removing grass thatch and creating seed beds may be necessary to allow natural colonisation on some sites, balanced against their impact on establishing trees (Woodland Trust, 2022). While recognising that trampling and rooting may create suitable conditions for germination, Forestry Commission (2021) recommend the exclusion of all grazing and browsing animals from most sites for up to several decades to allow establishment by natural colonisation. Murphy et al. (2022) recommend livestock exclusion for at least twelve years. Very low stocking densities may permit colonisation at some sites: this can be monitored and further reduced as necessary (Forestry Commission, 2021). On sites where domestic stock have prevented natural regeneration and reduced structural diversity, temporary exclusion of grazing for a few years followed by extensive cattle grazing may be most beneficial for woodland health (Mayle, 1999).

An appropriate stocking density can be estimated from: estimated forage productivity of each habitat in the grazed area, target utilisation rate for each habitat (see below), the amount of forage expected to be removed by wild deer and the species, season and duration of stock grazing, using an Excel tool included in the Woodland Grazing Toolbox (Scottish Forestry, 2024). Target utilisation rate is an expression of the percentage of plant material expected to be removed by herbivores and can be varied depending on the management objectives for a woodland. For woodland grazing: 10-15% is likely to result in low herbivore impacts; 20% is likely to result in medium herbivore impacts; 30% is likely to result in high herbivore impacts and 40% is likely to result in very high herbivore impacts. Grassland habitats often have up to 70% target utilisation, depending on the degree of agricultural improvement, so if a woodland grazing management unit includes areas of productive grassland and a woodland of low grazing potential, it could be reasonable to expect high utilisation of the grassland simultaneously with low (10-15%) utilisation of woodland forage. Such high utilisation of grassy habitats would maintain a short sward, reduce floral resources and might only be acceptable if the grasslands were of low conservation interest to begin with. The Woodland Grazing Toolbox provides further guidance on calculation of stocking densities for habitat mosaics. Any planned stocking density is an estimate of what is likely to be appropriate and the actual effects of grazing must be monitored. The Woodland Herbivore Impact Assessment (Armstrong et al., 2023) is appropriate for this and gives guidance on frequency and timing of monitoring. It may be supplemented by fixed point photography, repeated every five years or more often if rapid change is anticipated (Scottish Forestry, 2024).

Stocking density can be expressed in livestock units (LSU) to facilitate aggregation of different species and ages of animals. One LSU is the grazing equivalent of one adult dairy cow producing 3000kg of milk annually, without supplementary feed. Values for other bovines, equines, sheep, goats, pigs and rabbits are given by the (European Commission, 2023). The rewilding project on Knepp Estate uses 0.27 LSU per hectare (Tree and Burrell, 2023, cited in Lyons et al., 2024), compared to an average of 0.58 LSU (equivalent to one medium-sized suckler cow) per hectare across English lowland livestock farms (Natural England, 2009). Stocking density less than 1.2 sheep per hectare will allow most tree saplings to develop unbrowsed (Woodland Trust, 2022). Other sources recommend 0.05 LSU per hectare to allow regeneration in the presence of cattle grazing (Small, 2004). Hester *et al*. (1996) found more saplings of rowan, birch and ash in upland woods grazed at low intensity (0.6-1.2 sheep/ha) than medium (1.2-2.0 sheep/ha) or high (2.1-3.8 sheep/ha) intensities.

Laborde and Thompson (2015) found that stocking density of around four sheep per hectare for 5-6 months of the year with occasionally a few cattle prevented establishment of hazel *Corylus avellana* and hawthorn *Crataegus monogyna* on grassland adjacent to upland woodland in Cumbria. However, many stunted individuals were present which could rapidly establish scrub vegetation on relaxation of grazing pressure. Hazel was less palatable but also less resistant to grazing than hawthorn. Sheep are reputed to displace red deer, introducing the possibility that reduction in stocking density of sheep may fail to reduce browsing if deer return (Andrews et al., 2000).

Different grazing patterns can have different effects on tree damage and ecosystem services. “Extensive grazing” can be defined as a low stocking density of hardy breeds, either seasonally or year-round, with the aim of achieving conservation objectives. “Naturalistic grazing” is similar but year-round and with the aim of allowing natural processes to operate with minimal intervention rather than having a target state (Lyons et al., 2024). Others have defined naturalistic grazing as having no specified stocking density but requiring herbivore populations to be resource limited . “Pulse grazing” uses short durations of livestock access followed by a longer rest period, similar to “mob grazing” but at lower stocking densities. Pulse grazing is considered by some to replicate the seasonal movements of large herbivores in a (pre)historic landscape without fences more closely than extensive grazing, and perhaps to avoid depletion of floral resources across a landscape, although it reduces continuity of cattle dung availability for dung invertebrate communities. Short periods of high densities of herbivores should create niches for tree seeds to establish (Lyons et al., 2024). How To Rewild (How to Rewild, 2024d) recommend short durations (days to weeks or at most a few months) of low cattle stocking density (0.1 LSU per hectare). They report healthy tree regeneration at 25% of sites cattle-grazed for two months of the year compared to 5% of sites grazed year-round, although the stocking densities of these sites are not reported.

At Butcherlands Sussex Wildlife Trust reserve, very variable pulse grazing with cattle, with rest periods between 2-20 months and stocking density varying between 0-194 LSU days per hectare per year (average 86 LSU days per hectare per year i.e. LSU multiplied by the number of days grazed) has reduced tree and scrub regeneration. This does not wholly prevent scrub encroachment of open habitats so ultimate succession to woodland would be likely under this regime (Lyons et al., 2024).

A replicated study in upland ash, birch and rowan woods in Cumbria found more tree seedlings following summer grazing of sheep (8-17/100 m^2) compared to following winter grazing (4-6/100 m^2) (Hester et al., 1996). Generally, evergreen plants will be more browsed in winter and deciduous plants preferentially browsed in summer (Mayle, 1999).

Night-penning early in the growing season has been used in grazed orchards and contributes to the reduction of soil nutrient enrichment by grazing, leading to enhanced plant diversity (Paesel et al., 2019).

Contraceptives have been used to control goat populations at RSPB Inversnaid, although population control is more commonly carried out by shooting in feral populations or the slaughter of male yearlings in domestic herds (How to Rewild, 2024e).

Mayle (1999) gives recommendations of grazing regimes appropriate to different NVC woodland types, as well as practical considerations for the management of woodland grazing.

The Grazing Animals Breed Profiles Handbook, summarised in The Woodland Grazing Toolbox (Scottish Forestry, 2024), supplies opinion on the breeds least likely to derive their forage from woody species to assist breed selection. For cattle, hardy upland beef breeds and dual-purpose breeds are considered to browse more than lowland beef (particularly continental breeds) and dairy breeds (with the exception of Kerry cattle). For sheep, primitive (e.g. Soay, Shetland) and hill (e.g. Swaledale, Herdwick) breeds are considered to browse more than lowland breeds. However, all livestock breed differences are anecdotal and the total impact on tree resources is likely to be determined more by animal body size and hunger. How To Rewild (How to Rewild, 2024f) advise that Soay sheep will eat bramble and British Primitive browse preferentially on scrub so that 1-3 sheep of these breeds per hectare will convert scrub to grassland. Hardy breeds are considered less selective, taking poorer quality forage (Mayle, 1999). The most common breeds raised in woodland, in order of frequency, are Highland, Aberdeen Angus, Limousin, Shorthorn, Luing, North Devon and Welsh Black (How to Rewild, 2024d).

Supplementary feeding is usually supplied for welfare reasons before animals become so hungry that they resort to the extent of bark-stripping which would kill mature trees or shrubs (Lyons et al., 2024).

MODIFY ACCESS TO TREE RESOURCES BUT NOT AVAILABILITY

Conventional and electric fencing are considered to be effective for livestock, but tree guards and chemical repellents are not (Forest Research, 2024c). A Countryside Stewardship Grant may fund fencing to protect newly-planted trees from livestock: specifications and eligibility are described in (Rural Payments Agency and Natural England, 2024).

Fencing 4ft high with a single strand of barbed wire can be adequate to contain goats, provided that there is nothing on which goats can climb to jump over it. Electric fencing should be five strands of high tensile wire since electric tape fences are easily breached (How to Rewild, 2024e). Cattle grids can prevent goats crossing roads (How to Rewild, 2024e).

Spatial and temporal variation in grazing intensity is important to maintain the structural heterogeneity which benefits many invertebrate species. It can be encouraged by variation in availability of food, water, shelter and shade but in most practical situations is best achieved by temporary fencing or virtual fencing such as NoFence collars (Lyons et al., 2024; Woodland Trust, 2022). There is little evidence about the effects of virtual fence systems in woodland, but Forestry and Land Scotland are trialling the NoFence system to direct cattle grazing adjacent to a designated oakwood with the aim of creating niches among bracken and grass for oak regeneration. Cows wear GPS collars which emit a sound when the animal nears the programmed boundary, gradually increasing in pitch as the animal approaches and eventually delivering an electric pulse similar to that of a standard electric fence. A 15m boundary is required so the system is not suitable for small or narrow areas (How to Rewild, 2024f). QR codes linked to the virtual fencing allow the public to see the boundary of the grazed area and the animals' current location (Forestry and Land Scotland, 2022c). NoFence collars have also been used to control sheep grazing (How to Rewild, 2024f) and goat browsing (e.g. directing goats to clear unwanted scrub (Log et al., 2022)) although solar-powered collars may be impaired in winter or when the animals spend much time in shade (How to Rewild, 2024e).

Alley-cropping systems produce timber or fruit and beef/dairy in the same field using 24m wide grazing alleys between avenues of trees and are suitable for tree species which do not sucker into the field, demand a lot of water or poison livestock. Cattle are separated from trees by single-strand, high-tensile electric wire on wooden posts, adding a lower strand of electric wire if calves are present (How to Rewild, 2024d). Alley-cropping systems or traditional orchards are also used to graze sheep (How to Rewild, 2024e)

Temporary fenced exclosures may be used to allow establishment of browse-sensitive trees in grazed woodlands (Scottish Forestry, 2024). Two-year temporary exclusion of cattle and horse grazing significantly reduced mortality of planted English oak and ash seedlings in grassland-scrub mosaics (Jan Van Uytvanck et al., 2008). Around 70-80% of tree seedlings (depending on vegetation type) survived the third year after two years of exclusion.

Individual trees or small groups of trees in wood pasture systems can be protected against livestock by tree cages or fencing (Woodland Trust, 2022). Protection of individual trees against goats is challenging: exclosures around groups of trees or individual parkland tree guards may be successful if tall enough and constructed from strong, tight mesh held by strong stakes and providing a wide buffer zone around the tree(s). Tree tubes are ineffective as goats will push over staked trees to eat the leaves (How to Rewild, 2024e). Stakes and tree guards must be sturdy enough to withstand cattle and ponies rubbing against them (Mayle, 1999).

Retaining brash after conifer felling adjacent to a Scottish Atlantic oakwood protected seedlings against browsing by sheep and deer with the depth of brash affecting the probability of browsing. However, survival and performance of oak seedlings after two years was lower for seedlings planted in the brash than those planted in the clear area between brash lines, perhaps due to shading, acidity of the brash, water and nutrient limitation or limited root penetration (Truscott et al., 2004). The optimum depth and width of brash lines and suitable size and species for wood constituting the brash, remain to be determined.

Traditional herders describe a need to keep livestock slowly moving within forests to avoid overgrazing, trampling of saplings, bark-stripping and ruderalisation of the vegetation in resting areas (Varga et al., 2020). Traditional herding also creates variation between the patches frequently used, where the understorey becomes very open, and the less accessible places seldom used which retain a dense shrub layer.

Supplementary feed, water, salt licks and manure heaps should be kept at a distance from ancient and veteran trees (AVTs) to avoid excessive trampling and nutrient input. However, light or intermittent grazing is beneficial to maintain the parkland landscape in which these trees matured and reduce rank vegetation which shades lichens and insects on the trunk and may increase fire risk (Woodland Trust, 2021a). Alternative sources of shade for animals should be available and temporary fencing may be used to exclude livestock from AVT root zones, or slip rails included in fencing to allow occasional grazing access (Woodland Trust, 2021b, 2005). Fallen branches or other deadwood can be positioned around AVTs to protect from livestock and assist connectivity of deadwood habitats. Animals treated with veterinary medicines should be kept away from AVTs until the medicines are fully excreted (Woodland Trust, 2021a).

MODIFY TREE RESOURCE AVAILABILITY AND RESILIENCE TO DAMAGE

Sabre planting (see section on deer), in which 1.2-metre saplings are planted perpendicular to steep slopes, maintains the growing tips out of the reach of sheep (Woodland Trust, 2022). Sabre planting has been used on steep sites in Wales to protect against sheep, goats, cattle and roe deer; and in Scotland to protect against sheep, horses, roe and sika deer (Armstrong & Robertson, 2013).

For species such as willow which can grow from cut rods, planting willow pegs below ground level so that roots can establish before they sprout, or else 3 m willow forks whereby they are pushed into the ground far enough to prevent pulling out but have buds above browsing heights, have been suggested as approaches to encourage willow regeneration in the presence of browsers (Woodland Trust, 2022). Saplings of palatable species can be planted among existing thorny scrub to reduce access by browsers (Woodland Trust, 2022).

Associational resistance may allow sapling establishment at low stocking densities but not at higher densities when the rank vegetation or shrubs are themselves grazed and browsed (Smit et al., 2015). Associational resistance was demonstrated by Bakker *et al*. (2004) in broadleaved forest-scrub-grassland mosaics on floodplains in the Netherlands, Germany and New Forest grazed by cattle with or without equines and red deer (0.4-1.9 animals/ha). Young oaks were significantly associated with blackthorn *Prunus spinosa* although the reverse was true for mature trees, suggesting that blackthorn scrub facilitates the establishment of oak. The opposite pattern was observed for hawthorn *Crataegus monogyna* scrub with oaks and ash *Fraxinus excelsior* trees with blackthorn scrub. Survival of planted oak seedlings was significantly suppressed by grazing except among young blackthorn scrub, where oaks grew as well outside exclosures as within. Cattle had no impact upon the spread of blackthorn scrub.

Associational resistance against deer, sheep and cattle is considered to be provided by gorse *Ulex europaeus* plants nursery-grown for two years and clipped before planting within a few centimetres of birch seedlings. This is said to result in around 90% survival (French, 2012) although systematic studies are lacking.

Van Uytvanck *et al.* (2008) studied associational resistance provided to 6-month-old ash and English oak seedlings by tussocky *Juncus effusus*, tall *Carex acuta*, bramble thickets and ruderal pioneer vegetation against cattle and horses at 0.4-0.5 animals/ha. Tree survival after three years was greater in all of these situations (60-80%) than in open grassland (20-30%). In the context of wood pasture in Switzerland, 1- and 2-year-old Norway spruce saplings survived better in association with unpalatable plants yellow gentian *Gentiana lutea* and dwarf thistle *Cirsium acaule* under grazing by cattle (0.58/ha for 3-4 x 10-day periods in summer) and occasional horses. Survival was 33% in plots with unpalatable plants, 19% in plots where unpalatable plants had been cut and 14% in plots without unpalatable plants. Gentian was more effective than thistle and larger saplings survived better than smaller (Smit, Den Ouden, & Müller-Schärer, 2006).

Sweetbriar *Rosa rubiginosa* was used to provide associational resistance for saplings of silver fir, Norway spruce, sycamore and beech under exclusion, low intensity (83.6–106.1 LSU days ha^{-1}) and high intensity (181.7–204.1 LSU days ha^{-1}) of cattle grazing in the Swiss Jura Mountains (Vandenberghe et al., 2009). Deciduous saplings benefitted from nurse shrubs under both grazing intensities but the effect of nurse shrubs for conifers was not significant at high grazing intensity. Survival of deciduous trees planted close to nurse shrubs was not significantly different than under grazing exclusion. Associational resistance was stronger under lower grazing pressures and conifers were less resistant to grazing overall.

Woodland managers surveyed by Armstrong (2003) noted that birch, beech and oak regeneration is confined to the shelter of bracken, bramble, holly and fallen deadwood in cattle-grazed woodlands.

Oak, aspen and willow are favoured by cattle while hawthorn, holly, rowan, birch and alder (in descending order of avoidance) are not (How to Rewild, 2024d). Hawthorn and field maple are considered less susceptible to bark-stripping by goats than other native broadleaves (How to Rewild, 2024e). Sheep are thought to prefer Silver Birch, Willow, Hazel and Alder (How to Rewild, 2024f)

Movement of livestock within a wood will be influenced by water availability, dense scrub, boggy ground, watercourses, crags and ravines so (relatively) inaccessible areas should be excluded when calculating the area available for grazing and determining stocking density (Scottish Forestry, 2024).

5.1.8 Horses and ponies (Family Equidae)

5.1.8.1 Background and ecosystem services

Horses and ponies are increasingly being reintroduced to former agricultural areas across western Europe to restore natural and diverse habitats (Hester et al., 2000; J. Van Uytvanck et al., 2008). In the UK, there are an estimated 847,000 horses and ponies (Furtado et al., 2022), including rare native breeds such as Exmoor, Dartmoor, Fell, Highland, and Eriskay ponies, which have been used for conservation grazing (Wildlife and Countryside Link, 2020). Wild horses and ponies are animals of a range of habitats, but they most commonly occur in high-quality grasslands and are only rarely found in closed-canopy woodlands (Scottish Forestry, 2024).

5.1.8.2 Damage and impacts

The British Horse Society recommends 1–1.5 acres per animal for healthy levels of grazing (cited in Furtado et al., 2022), but stocking densities are often many times higher than this (How to Rewild, 2024g). Horses and ponies are selective grazers, choosing to eat certain plants while ignoring others. This browsing behaviour can result in vegetation mosaics, with closely grazed patches interspersed among areas of undisturbed vegetation (How to Rewild, 2024g; Spencer, 2023; Wildlife and Countryside Link, 2020). Despite their potential conservation role, horses and ponies can cause significant damage to trees and pastures if not managed properly. Horses and ponies may also rub against trees (Scottish Forestry, 2024) or uproot newly-planted trees (Forest Research, 2024b). High stocking densities and overgrazing can lead to excessive soil damage from trampling (Furtado et al., 2022). This, in combination with the introduction of seeds from supplementary food, can lead to the overrepresentation of often invasive weed species such as nettles, dock, thistles, and buttercups (Furtado et al., 2022).

5.1.8.3 Identification of damage

Although horses and ponies do not browse on woody species as often as other mammals, they may occasionally do so, particularly when other food sources are scarce. In these instances, they can cause severe damage to shrubs and trees by stripping bark up to a height of 2.5 m, depending on the size of the individual animals. Such damage is identifiable by the diagonal incisor marks left by both jaws (Forest Research, 2024b; Forestry Commission, 2023; Scottish Forestry, 2024). How to Rewild (2024g) notes that a pasture is considered as overgrazed when the sward height is less than 5 cm.

5.1.8.4 Protection against damage

Tree protection methods identified in the literature for horses and ponies are shown in Figure 15, with details and evidence base synthesised below.

MONITORING BEFORE DAMAGE

The Woodland Herbivore Impact Assessment Method (Armstrong et al., 2023) can be used to assess the impact of horses and ponies on woodland and potential woodland habitats. By evaluating recent trampling, browsing, or grazing effects on various plant indicators, this monitoring method can help predict future changes in habitat structure and species composition under current conditions (Scottish Forestry, 2024).

MODIFY ACCESS TO TREE RESOURCES BUT NOT AVAILABILITY

Both agricultural stock fencing and electric fencing are considered effective for protecting trees from horses and ponies, though a 1.5 m buffer zone between the fence and the trees is required (Forest Research, 2024b, 2024c). Rotational grazing, seasonal grazing and reducing the overall levels of grazing have been identified as possible management options (Mitchell & Kirby, 1990). A low level of grazing can provide a greater diversity in vegetation structure and species composition, when compared to high levels or grazing or the absence of grazing in fenced woods (Mitchell and Kirby, 1990). Tree guards are not viable for individual tree protection, and chemical repellents are not legally permitted for use against horses and ponies in the United Kingdom (Forest Research, 2024b, 2024c). How to Rewild (2024g), mention that trees planted in pastures can be protected by using parkland guards, but there is no empirical evidence for our search to support this. In Europe, contraceptive darts have been used on small, feral horse populations, particularly in areas where culling is publicly opposed or illegal. However, a study conducted in Romania concluded that while contraception of feral horses is feasible, the approach is time consuming and resource intensive (Rosu et al., 2016). How to Rewild (2024g) advises against providing supplementary feed, as the additional nutrients in the feed can increase nitrogen levels in the soil, through faeces, and negatively impact plant diversity. Olff et al. (1999) also note that supplementary feed should not be provided as doing so may maintain population densities at unnaturally high levels, and hence potentially increase grazing pressure.

Olff et al. (1999) suggest that dead trees should not be removed, and that unpalatable forbs and shrubs should be allowed to establish, with both strategies likely to act as natural protective barriers for more palatable tree species. However, the Wildlife and Countryside Link (2020) highlights that ponies native to the United Kingdom are highly adaptable foragers. For example, they report that New Forest ponies have been observed consuming large quantities of bracken once its toxicity decreases, with no apparent ill effects (Wildlife and Countryside Link, 2020). Similarly, both Fell and Highland ponies are known to graze on reeds and rushes year-round. During the winter, ponies have also been seen uprooting plants like nettles (Wildlife and Countryside Link, 2020). Moreover, Gilbert (1980) recorded that horses can kill juniper *Juniperus spp.*, which is unpalatable, by gnawing at the bark (cited in Thomas et al., 2007). Thus, empirical evidence of horses and ponies avoiding unpalatable foods is lacking.

MODIFY TREE RESOURCE AVAILABILITY AND RESILIENCE TO DAMAGE

There is some evidence pollarding may help mitigate damage to mature trees by horses and ponies. Rackham (1980), as cited in Mitchell and Kirby (1990) describes examples in England where trees were cut to about 2 m above ground level so that the regrowth would be out of reach of browsing animals. While horses and ponies were not explicitly mentioned, the technique is likely a viable option to reduce grazing.

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