Qualitative impact assessment of land management interventions on Ecosystem Services

Report-3 Theme-5D: Biodiversity - Integrated System-Based Actions

30-June-2023

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This report is one of a set of reviews by theme:

Braban, C.F., Nemitz, E., Drewer, J. (2023). *Qualitative impact assessment of land management interventions on Ecosystem Services ("QEIA").* Report-3 Theme-1: Air Quality (Defra ECM_62324/UKCEH 08044)

Birnie, J., Magowan, E., Law, R., Lucas, O.T., Hassin, A.E.J. (2023). *Qualitative impact assessment of land management interventions on Ecosystem Services ("QEIA").* Report-3 Theme-2: Greenhouse Gases (GHG) (Defra ECM_62324/UKCEH 08044)

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Williams, J.R., Newell Price, J.P., Williams, A.P., Bowes, M.J., Hutchins, M.G. & Qu, Y. et al. (2023). *Qualitative impact assessment of land management interventions on Ecosystem Services ("QEIA")*. Report-3, Theme-4: Water (Defra ECM_62324/UKCEH 08044)

Staley, J.T., Botham, M.S., Broughton, R.K., Carvell, C., Pywell, R.F., Wagner, M. & Woodcock, B.A. (2023). *Qualitative impact assessment of land management interventions on Ecosystem Services ("QEIA")*. Report-3 Theme-5A: Biodiversity - Cropland (Defra ECM_62324/UKCEH 08044)

Keenleyside, C.B. & Costa Domingo, G. (2023). *Qualitative impact assessment of land management interventions on Ecosystem Services ("QEIA")*. Report-3 Theme-5B: Biodiversity - Grassland (Defra ECM_62324/UKCEH 08044)

Maskell, L. & Norton, L. (2023). *Qualitative impact assessment of land management interventions on Ecosystem Services ("QEIA")*. Report-3 Theme-5C: Biodiversity - Semi-Natural Habitats (Defra ECM_62324/UKCEH 08044)

Siriwardena, G.M. (2023). *Qualitative impact assessment of land management interventions on Ecosystem Services ("QEIA")*. Report-3 Theme-5D: Biodiversity - Integrated System-Based Actions (Defra ECM_62324/UKCEH 08044)

Bentley, L., Feeney, C., Matthews, R., Evans, C.D., Garbutt, R.A., Thomson, A. & Emmett, B.A. (2023). *Qualitative impact assessment of land management interventions on Ecosystem Services ("QEIA")*. Report-3 Theme-6: Carbon Sequestration (Defra ECM_62324/UKCEH 08044)

Short, C., Dwyer, J., Fletcher, D., Gaskell P., Goodenough, A., Urquhart, J., McGowan, A.J., Jones, L. & Emmett, B.A. (2023). *Qualitative impact assessment of land management interventions on Ecosystem Services ("QEIA")*. Report-3.7: Cultural Services (Defra ECM_62324/UKCEH 08044)

A list of all references used in the reports is also available as a separate database.

Foreword

The focus of this project was to provide a rapid qualitative assessment of land management interventions on Ecosystem Services (ES) proposed for inclusion in Environmental Land Management (ELM) schemes. This involved a review of the current evidence base by ten expert teams drawn from the independent research community in a consistent series of ten Evidence Reviews. These reviews were undertaken rapidly at Defra's request and together captured more than 2000 individual sources of evidence. These reviews were then used to inform an Integrated Assessment (IA) to provide a more accessible summary of these evidence reviews with a focus on capturing the actions with the greatest potential magnitude of change for the intended ES and their potential co-benefits and trade-offs across the Ecosystem Services and Ecosystem Services Indicators.

The final IA table captured scores for 741 actions across 8 Themes, 33 ES and 53 ES-indicators. This produced a total possible matrix of 39,273 scores. It should be noted that this piece of work is just one element of the wider underpinning work Defra has commissioned to support the development of the ELM schemes. The project was carried out in two phases with the environmental and provisioning services commissioned in Phase 1 and cultural and regulatory services in a follow-on Phase 2.

Due to the urgency of the need for these evidence reviews, there was insufficient time for systematic reviews and therefore the reviews relied on the knowledge of the team of the peer reviewed and grey literature with some rapid additional checking of recent reports and papers. This limitation of the review process was clearly explained and understood by Defra. The review presented here is one of the ten evidence reviews which informed the IA.

Acknowledgments

This project work and the resultant reports and databases were made possible by funding from the Department of Environment, Food and Rural Affairs, under contract ECM_62324. UKCEH and all the project participants are very grateful for the support we have received from DEFRA colleagues. In particular we would like to thank Tracie Evans, Hayden Martin, Daryl Hughes, Chris Beedie and Catherine Klein for their support and constructive inputs to the exercise. We would also like to thank our numerous external contributors and reviewers, some of whom have chosen to remain behind the scenes, and we are very grateful for the expansive and meticulous body of peer-reviewed evidence our authors have been able to refer to and make use of.

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1 INTRODUCTION

1.1 CHALLENGES WITH REVIEWING EVIDENCE FOR 'BIODIVERSITY'

Evidence for biodiversity effects of actions is particularly complex: they are frequently intended to benefit multiple taxa and all individual species could respond differently, so hypothetical impacts and data can show multiple different directions of response, simultaneously. Scaling may also be critical, and variable, with some responses only occurring beyond a given scale of implementation and others being dependent on or impacted by habitat context; scaling often cannot be assumed to be linear. Further, evidence can take different forms, such as on-off action comparisons of presence, abundance, species richness, activity or cover, or effects of intervention area or quantity on population trend. These metrics can also be measured at the scale of the action (within fields), between fields, between farms or across larger spatial survey units, such as 1km gird squares. The availability of ongoing, independent monitoring data recording these metrics mean that evaluation of the impact of management interventions in practice, in real schemes and at the landscape scale, is available for some groups, notably birds, but evidence is limited to small-scale trials and experiments under controlled conditions for other taxa. Evidence may therefore relate to implementation in practice or evaluation of the general principles behind an action, or both, and it is arguable which is more valuable in principle.

Further to the above, some taxa are easier to monitor than others and some have received more research and monitoring investment, or have more existing data resources that can be mined. In addition, the ecology of habitat relationships and critical life stages for populations is better known for some species or groups than for others: actions that address limiting factors will be more effective in terms of population impacts than those that simply drive habitat selection or redistribution, for example, but knowledge and data may only support the latter for many groups. The mobility of taxa will also affect their responses to actions and ease of monitoring. Plants are easy to monitor but may be responding to historical rather than current management because of time lags in their ability to respond to current conditions (particularly for long lived plants).

All of the above means that there is much more evidence, with more studies and better data quality, for birds and butterflies, compared to small mammals and carabid beetles, for example. In turn, this means that there is more chance that both positive and negative, or neutral, effects of interventions will have been identified for some taxa than others. Where data quality is higher, for example allowing sensitive, long-term evaluations of effects on population change, as opposed to just spatial comparisons of occurrence, it is more likely that subtle, perhaps less predictable effects (and, indeed, spurious correlations) will be found. Such data may allow the identification of a lack of a national-scale effect despite the occurrence of field-scale benefits, for example. Further, monitoring of management within schemes is more likely to identify mixed effects than controlled trials, because there will be more noise from uncontrolled environmental variation, influences from sub-optimal implementation by farmers and less targeted taxon sampling. Overall, the resulting hierarchy of evidence quality, spread across multiple species and broader taxonomic groups, makes a rounded, integrated, evidence-based and certain conclusion in respect of effects on biodiversity as a whole next to impossible to achieve, even for well-studied interventions.

1.2 SCOPE OF BIODIVERSITY UNDER CONSIDERATION

Here, we have focused on the key groups at which actions are aimed and the best available evidence for those taxa. We have not considered soil or aquatic biodiversity. The quality of that best evidence varies hugely between actions and taxa, so ratings are not necessarily comparable between actions: the evidence may be equally relatively positive for two actions, but with very different absolute levels of certainty and numbers or scale of studies. Many groups (e.g. bats) have never been studied with respect to a range of actions that might affect them – evidence has been flagged where we are aware of it, but we have not highlighted all the cases where evidence is absent for specific groups but would ideally exist (or be better). Evidence ratings refer to the balance of the available evidence, across the range of groups for which evidence exists, considering the certainty/consistency of the patterns that have been found. Where multiple taxonomic groups could respond to a particular intervention, but it is primarily aimed at one group, the evidence for the latter group is given greater weight in scoring.

1.3 Challenges with action definitions

Broad option definitions have not helped in the review process – especially for specifics such as magnitude and timescale, which are always variable. Non-specific actions rarely match well to the interventions that have actually been assessed in trials or implemented in previous schemes and then monitored. This means that certainty of effect is generally lower than would have been the case with more specific categories, such as lists of specific options that have been implemented in Countryside Stewardship. Collecting multiple actions, such as '**ETPW-200x** Provide nesting and roosting sites (e.g. fallow plots/areas for ground nesting birds and invertebrates)', include such a wide range of actual individual actions and potentially responding species, each of which combinations could have its own evidence base, that a confident summary is not possible.

Evidence reviews by action do not facilitate the comparison of different alternative options with respect to particular targets or contexts, e.g. to find the best way to benefit a particular target or the best management to apply in a particular location. As an example, there are multiple actions for the management of field margins, and those are often clearly divisible by different fine details of possible management, so it would be valuable to be able to compare options directly, but this is not aided by the structure of this process.

None of the actions here are specific to certification schemes supported by those within the industry to enable them to market goods under particular labels, such as organic or pasture-fed products. These schemes largely involve practices that may be highly beneficial to the delivery of ecosystem services from farmed land (not least because they include minimal inputs). Given the extensive evidence base for the benefits of such systems (in particular 'organic' farming) for biodiversity, the omission of individual actions specific to these systems is significant.

The titles of many actions were ambiguous, so we have written our interpretations in the text for each section; we have then reviewed according to those interpretations. Again, more precise names, or descriptions of single management actions, such as would be used in an agri-environment prescription, would have simplified the review process and would have made the results more easily interpretable.

1.4 Evidence scoring categories

Much of the evidence regarding agri-environment effects and many of the options found in current schemes concern species of various taxa that are found primarily in agricultural land. Many of these are also priority species. Many species in these habitats also provide key ecosystem services such as pollination and pest control, and/or have declined in recent decades, even if they are not (yet) on the red list.

'Condition' of priority habitats, such as in SSSIs, is a well-established concept, but 'condition' of agricultural land tends to refer to suitability for agriculture, as reflected in the acronym GAEC, for 'Good Agricultural and Environmental Condition'.

We have brought the wider biodiversity gap and agricultural land condition uncertainty (from a biodiversity perspective) together, by considering 'condition of agricultural land' to refer to 'abundance and species richness of wider biodiversity'. Similarly, for consistency, references to 'condition of semi-natural habitat' are considered to include 'abundance and species richness of species typical of that habitat'. This has allowed us to consider the breadth of available evidence, reflecting historical and contemporary conservation concern, as far as is possible.

2 OUTCOMES

Various biodiversity benefits will be provided, as collected in the services collected above as Biodiversity adaptation under a changing climate and Maintaining habitats, nursery populations (and other stages of life cycles) [semi-natural habitats and priority species in productive farmland]. Habitat buffering options (3.1.2.1 and 3.1.2.2) would benefit Nutrient pollutants and perhaps Water Quality. Management of invasive species (3.1.1.1) would obviously be relevant to the Invasive Non-Native Species service, as would biosecurity options (3.1.3.2 and 3.1.3.4) and control of damaging species (3.1.3.5).

3 MANAGEMENT BUNDLES

3.1 BUNDLE: SYSTEMS ACTION

Systems actions with biodiversity benefits consist of interventions that cut across farming systems and semi-natural habitat types, or that are applied at the whole-farm or landscape level. The outcomes are intended to involve various aspects of biodiversity, some that with a conservation focus (populations or occurrence of species that have been identified as being targets) and some more functional, such as pollinators or predators of crop pests. Where there are habitats that depend upon low-intensity agricultural management, or priority species that depend on the use of specific forms of agriculture, or the scale of definition of habitats is small (such that hedgerows, for example, are considered as a habitat type), the management will also contribute to maintaining habitats.

3.1.1 ECPW-078: Monitor and control damaging terrestrial plant species

This action is assumed to relate to the management of invasive, terrestrial, plant species (native species can be damaging to agricultural interests, but the definition of 'damaging' impacts of native species in a semi-natural context is problematic). 'Damage' is assumed to refer to the stability of native wildlife communities and the abundance of component species. The 'action' must actually involve multiple quite different physical or chemical interventions, depending on the invasive species and environmental context, each with its own evidence base. Few UK control or eradication programmes have been successful; early identification of invasive status and prediction of effect would aid this by supporting action when invasives still have limited distributions (Manchester & Bullock 2000). The GB Non-native Species Secretariat (NNSS) addresses this by risk assessment, risk management and subsequent prioritisation processes leading to action/contingency plans for identified problems (http://www.nonnativespecies.org/). Notable invasives considered by NNSS include *Cotoneaster* spp. (Cotoneaster), *Fallopia japonica* (Japanese Knotweed), *Heracleum mantegazzianum* (Giant Hogweed), *Impatiens glandulifera* (Himalayan Balsam) and *Rhododendron ponticum* (Rhododendron). These species have a range of impacts, dependent upon their habitat preferences and growth habitats, but generally involving out-competition of native species and the alteration of habitat structure. The NNSS collates species-specific management information and evidence for the effectiveness of different measures. It also includes action plans for some species, although currently not for any terrestrial plant species.

Techniques to control these species can involve a range of methods including mechanical control such as cutting or cultivation, or herbicide application (Environment Agency 2010). However, the breadth of potential actions that are subsumed within this category make a full review unfeasible here: the range is too large.

There are potentially multiple interventions per invasive and a separate review could be conducted for each (e.g. Tyler & Pullin 2005). Do commonly used interventions effectively control Rhododendron ponticum? Systematic Review No. 6. Centre for Evidence-Based Conservation, Birmingham, U.K.. Only general issues with this broad class of intervention are, therefore, considered below.

3.1.1.1 Causality

Control, i.e. reduction or removal, of damaging plants would have positive effects on native species by definition, provided that recovery of native species is possible. If propagules of the latter are no longer present, additional reintroduction measures or other facilitation may be required, however, and such measures may increase the speed of recovery in any event. The issue with causality here is that actions for any given invasive species in a given context need to be shown to be effective in reducing prevalence and in sustaining that low level. This will inevitably be variable between interventions. As an example, Pysek et al. (1995, 2007) showed that removing all umbels of giant hogweed plants at peak flowering time dramatically reduced seed production in the species, but no other relevant studies were found by Dick et al. (2015).

3.1.1.2 Co-Benefits and Trade-offs

Agricultural production may benefit from control of invasives.

[TOCB Report-3-6 Carbon **ECPW-078**] The removal of woody vegetation is associated with the loss of vegetation carbon stocks and potentially significant soil carbon losses, depending on the method of removal and extent of soil disturbance (Matthews et al., in prep.). Invasive plants are often associated with high rates of primary productivity, potentially enhancing rates of carbon sequestration. However, litter produced by invasive plants may be more decomposable than native littler on average, may reduce litter from native species leading to a net decrease in litter, and in some cases can significantly increase the risk of fire and associated carbon losses (Peltzer et al., 2010; Robertson & Coll, 2019). Furthermore, the removal of invasive species could facilitate the establishment of native vegetation with equivalent or higher sequestration potential, although this may years or decades in the case of woodland. It has also been suggested that invasive plant biomass could be used to contribute to biochar production, although this research of this possibility is at an early stage (Feng et al., 2021). In the case of invasive species removal, there may be significant other considerations that override contributions to carbon sequestration (Lindenmayer et al., 2012). If vegetation removal is carried out using livestock, any GHG emissions associated with this practice should also be considered.
*[Duplicated or related evidence: **EBHE-301**: Control invasive plant species by chemical means to help manage archaeological sites; **ETPW-146** Control bracken and scrub by mechanical means; **ETPW-272** Control bracken and scrub by targeted grazing and trampling; **ETPW-262** Control invasive plant species by chemical means to help manage and restore habitats; **ETPW-268**: Control invasive plant species on or around Scheduled Monuments by chemical or mechanical means.; **ETPW-263** Weed wiping, precision or spot spraying to control injurious weeds and invasive plants to help manage habitats]*

Food and fibre production	Area under production or yield and outside of ELM	**N**
Global, regional & local climate regulation	Above ground carbon sequestration	**LTD***
	Below ground carbon sequestration	**LTD***

3.1.1.3 Magnitude

This will vary between invasive species, specific actions and contexts.

3.1.1.4 Timescale

Timescales are likely to vary from a few years to decades: removal action can be rapid, but recovery, even with reintroduction of native species and facilitation such as fencing or tree guards, is likely to be limited by the speed of ecological processes and vegetation growth.

3.1.1.5 Spatial Issues

Targeting of this action to areas where either the damaging species is causing most damage or native species have the best chances of recovery (e.g. close to source populations) will increase effectiveness.

3.1.1.6 Displacement

N/A

3.1.1.7 Maintenance and Longevity

Removal of damaging species is unlikely often to be a discrete, single activity; it is likely that ongoing control to prevent re-establishment will be necessary, albeit with a lower level of effort. In the pre-control stage of monitoring for occurrence or impact, that is an ongoing, indefinite activity, by definition.

3.1.1.8 Climate Adaptation or Mitigation

The pressures driving the problems associated with damaging species could be exacerbated or reduced by climate change, depending on the characteristics of the invasive species. However, novel climate conditions are likely to facilitate expansion of further invasive species, leading to new challenges for monitoring and control.

3.1.1.9 Climate Factors / Constraints

This action could contribute to climate change mitigation if the spread of invasive species is part of the climate change impact. As per 3.1.1.1.8, the range and difficulty of measures required to control invasives could increase with climate change.

3.1.1.10 Benefits and Trade-offs to Farmer/Land manager

If invasives impact on production as well as semi-natural habitats and priority species, this action could be beneficial to farmers as well as to priority species and habitats.

3.1.1.11 Uptake

N/A

3.1.1.12 Other Notes

N/A

3.1.2 ETPW-271: Create/manage/enhance buffer strips to encourage natural predators and species diversity

Buffer strips around fields can have multiple functions, including protection of watercourses from run-off (**ECPW-042**), protecting adjacent (field boundary other habitat) vegetation from spray drift (**ETPW-108**) and provision of resources in uncropped habitat for multiple taxa (**ETPW-205C**), but also including a function as refugia for the natural enemies of crop pests. In the latter way, the buffers function similarly to beetle banks (**ETPW-207**). Unlike sown grass buffers that simply protect adjacent habitat, buffers that aim to enhance biodiversity feature diverse vegetation composition and structure, such as to provide food and micro-habitat resources for a high diversity of invertebrates, in particular. The margins are not fertilised and only spottreated with herbicides if necessary. This review focuses on grass buffers, established by sowing and/or natural regeneration. Cultivated margins for rare arable plants (e.g. as a sub-option in option R3 in the Countryside Stewardship Scheme (Defra 2003) are a separate action, falling within the broad range of actions within **ETPW-205C**, as are low-input cropped margins – see **ETPW-240**. Note that many studies will not have distinguished between these forms of action.

The specific addition for **ETPW-271** is for 'natural predators', which implies an ecosystem service role, although this is not spelled out. No attempt has been made here to review the evidence for pest predation ecosystem service function, per se, although some relevant literature has been found, i.e. scoring considers the invertebrates, not the ecosystem service. The specific addition for **ECPW-042** is a 'riparian' context, i.e. emphasising the run-off prevention element, but aside from a presumed spatial context difference from being close to a watercourse, biodiversity relationships should be the same as for **ECPW-038** Create/ manage/ enhance buffer strips.

Enhancement consists of adding plants providing more structure and floral and/or seed resources to a default sown grass strip (Vickery et al. 2009).

3.1.2.1 Causality

3.1.2.1.1 General margin effects

The biodiversity benefits of margin buffer strips have been reviewed extensively by the Conservation Evidence project (www.conservationevidence.com, Dicks et al. 2020). They found that studies from across Europe indicated that planting grass buffer strips (some margins floristically-enhanced) increased arthropod abundance, species richness and diversity, including specific mention of bumblebees. Grass buffers also benefited bird numbers, densities, species richness and foraging time, plant cover and richness, and small mammal activity and numbers. In other studies, grass buffers have bene shown to benefit reptiles and amphibians (Salazar et al. 2016).

However, of three bats considered by McHugh et al. (2018), Nathusius' pipistrelle activity was negatively correlated with the presence of grass margins. Further, Dicks et al. (2020) found a number of studies showing that planting grass buffer strips had no clear effect on insect numbers, bird numbers, invertebrate pest populations or arable plants, but fewer studies than showed some benefits. Further, at a larger scale, two replicated site comparisons from France and Ireland found farms with buffer strips did not have more plant species than farms without strips (Dicks et al. 2020).

In the specific case of riparian buffer strips, Dicks et al. (2020) found three studies showing increased diversity or abundance of plants, invertebrates or birds and greater cover of vegetation associated with water vole habitats.

Overall, there appear often to be positive effects across taxa, although not all species or groups that might nominally benefit do so. This may relate to scales of evaluation, i.e. local or short-term positive associations can commonly be found, but sustained or landscape-scale effects are less detectable. However, there is little evidence of any negative effects.

3.1.2.1.2 Types of margin management/enhancement

As well as the creation of margin buffers, how they are enhanced is critical to the benefits provided to various groups. Overwintering of soil dwelling arthropods and especially carabid beetles has been found to be much higher in unmown perennial field margins than in mown grass strips or barren crop fields (van Alebeek et al. 2006). Insect abundance in general is increased by structurally more diverse, enhance margin swards, with perennial forbs and management involving reduced chemical inputs, cutting and cultivation having particular positive effects (Vickery et al. 2009). Margin strips enhance soil macrofauna relative to cropped fields, but management by scarification has negative effects, so may show antagonistic effects relative to enhancement for other groups (Smith et al. 2007). In a grassland management context, fenced only, rotavated and reseeded (with a grass and wildflower mixture) margin treatments increased on the abundance and taxon richness of five arthropod trophic groups (detritivores, herbivores, predators, parasitoids and hyperparasitoids) in pastoral farmland (Anderson et al. 2007). Beyond sown grass buffers, naturally regenerated unsprayed field margins were more attractive to foraging bumblebees and honeybees than cropped field margins managed as conservation headlands (Kells et al. 2001), so choice of margin management approaches may benefit from a wider comparison – see **ETPW-205C** and **ETPW-240**.

The margin action types that are based on improving habitat quality by enhancing floral resources are those that are studied most often for impacts on pollinating insects. Margins supplemented with sown wildflower seed mixes, or pollinator-specific mixes, were found almost always to have a positive effect on the local abundance and species richness of pollinating insects (Staley et al. 2016). Several studies of bumblebees compared multiple options not only against a crop control, but also against each other, in general showing that whilst margins that had not been sown with a flower seed mix (grass buffer strips, uncropped margins and conservation headlands) had greater pollinator abundance or species richness than crop edges, sown flower margins most often outperform these other options. Further, studies including a measure of flower abundance show that significant effects of wildflower margins are due to increased floral resources (Staley et al. 2016).

Sown margins need to be mown periodically to maintain a sward and to prevent scrub encroachment. Scarification, where practised, might need to be annual and flower densities are likely to be maximised by particular mowing or scarification regimes. The value of margins for insect food provision for farmland birds is maximised by regular mowing of half of the margin vegetation at a time, hence providing both insect abundance and availability (Douglas et al. 2009).

3.1.2.1.3 Natural enemies and pest control

In the specific area of support for pest natural enemies and the ecosystem service of pest control, Ramsden et al. (2015) showed that margins providing floral resources and, to a lesser extent, aphid prey were more effective than simple grass margins. Comparing different crop types and pest species relevant to those crops, in the context of the effects of the provision of winter refuges to support natural enemy densities; some pest-crop combinations showed positive effects, but others were negative (van Alebeek et al. 2006). Surveys of twelve winter wheat fields with varying densities of surrounding field margins showed negative correlations between each of the total number of predators and Cantharidae and field margin density at local scales, but predatory Staphylinidae, especially Tachyporus spp., showed a positive correlation with field margin density at larger scales (Oaten et al. 2008). Holland et al. (2008) measured both natural enemy numbers and effects on cereal aphid pests from 6m margins around fields (versus 2m margins), separating the effects of flying and epigeal predators. There was no evidence that the wide field margins increased natural enemies within the adjacent field. The wide field margins were considered to have no benefit for biocontrol because flying predators capable of moving between fields were primarily responsible or the amount of uncropped land suitable for natural enemies was not a limiting factor in the landscape. However, natural pest control of wheat aphids is enhanced near flower-enhanced field margins and natural pest control declines with distance from the crop edge (Woodcock et al. 2016).

3.1.2.2 Co-Benefits and Trade-offs

Soil protection and water quality should be enhanced, dependent upon buffer location and orientation relative to crops, although management by scarification may reduce value in this regard.

[TOCB Report 3-3 Soils **ETPW-271**] Establishing vegetative cover where there may have been no growing cover in previous circumstances or establishing more permanent vegetative covers is likely to reduce soil erosion extent and result in some positive benefit for soil organic matter and structure.

[TOCB Report-3-5A Croplands **ETPW-271**] The effects of **ETPW-271** on Cropland biodiversity will be similar to those of **EBHE-117** Create/ enhance/ manage contour grass strips and other grassy strips and field margins.

See full review of **EBHE-117** under the Grass margins, strips and corners grouping for more detail on the Croplands biodiversity evidence (in Report-3-5A Croplands).

RAG rating for specific ecosystem service:
GREEN** maintaining species / wider biodiversity
GREEN* presence of rare or priority species
AMBER TL** pollination

3.1.2.3 Magnitude

Variable dependent on margin enhancement and focal taxon. Effects within fields fall with distance into field centres, so it is likely that many effects will be rather local and require high densities of margins to be significant.

3.1.2.4 Timescale

Effects will probably appear within a year or two for most taxa, as they require vegetation growth, but the vegetation will develop quickly.

3.1.2.5 Spatial Issues

Effects within fields fall with distance into field centres, so it is likely that many effects will be rather local and require high densities of margins to be significant. There is some evidence that a combination of field margin buffers and in-field uncropped patches for skylarks in the same fields has a negative effect on the performance of the latter (Morris & Gilroy 2008).

3.1.2.6 Displacement

Margins should have minimal displacement effects because they are placed in parts of fields where crop productivity is typically low and economic losses are minimal. ‘Squaring off’ field edges may also actually increase the efficiency of farm operations.

3.1.2.7 Maintenance and Longevity

Margins need at least approximately annual management, e.g. by mowing, to maintain habitat condition. Floral diversity is likely to be enhanced by re-sowing every few years.

3.1.2.8 Climate Adaptation or Mitigation

Margins increase landscape connectivity and hence may enhance movements in response to climate change for less mobile taxa.

3.1.2.9 Climate Factors / Constraints

None

3.1.2.10 Benefits and Trade-offs to Farmer/Land-manager

Ecosystem service benefits of pest control and pollination may improve yields and/or reduce the need for inputs.

3.1.2.11 Uptake

N/A

3.1.2.12 Other Notes

None

3.1.3 ETPW-108: Buffer priority habitats

This suite of actions could refer to sown grass buffer strips, shelter belts or ‘sacrificial’ areas of the target habitat that are established around cores of priority habitat. Buffer strips around fields can have multiple functions, including protection of watercourses from run-off (see **ECPW-042**), provision of resources in uncropped habitat for multiple taxa (see **ETPW-038**) and resource provision for the natural enemies of crop pests (see **ETPW-271**), but also including a function protecting adjacent semi-natural habitats and field boundary vegetation from spray drift. Vegetation composition and structure may have limited impacts on the effectiveness of this type of buffer. Shelter belts (see **ECAR-033C**) would protect from prevailing winds or harsh weather. Here, it is assumed to refer to ‘sacrificial’ buffer zones that could both soften hard habitat edges (see **ETPW-109** and **ETPW-213C**) and protect the core habitat area by absorbing deleterious edge effects from surrounding agriculture, and other pressures.

3.1.3.1 Causality

This concept has a long history of application at the landscape scale, aiming to buffer major protected areas such as biosphere reserves (Bennett & Mulongoy 2006). Buffering involves managing the area surrounding a wildlife site in ways which reduce adverse effects on the site, and sustain positive landscape interactions (Jongman & Pungetti 2004), with particular potential benefits for small, isolated sites (Lawton et al. 2010). However, it appears that there has been little research or monitoring of the functioning of buffers in practice, or comparing the effectiveness of different buffer types or widths for preserving the intactness of the target habitat, or preventing the ingression of undesirable biotic or abiotic influences. Further, clearly, this action could encompass a broad range of specific management actions and contexts (e.g. different priority habitats and different external threats) and it is quite likely that the buffer effectiveness will vary considerably as a result. The general principle seems incontrovertible: where habitats are under pressure from external factors beyond their borders, a buffer that restricts the influence of those factors by reducing passage or absorbing impact can only have a positive effect (assuming that isolation of the target habitat patch is not increased), and the wider the buffer and the more similar it is in habitat structure, the greater the effect will be. Examples of habitat degradation from the types of external influence concerned here constitute evidence of the value of the approach.

Buffering priority habitats was considered as a high-level management option for enhancing landscape adaptation to climate change by Lawton et al. (2010). They viewed it as contributing to wider environmental improvement within the general context of establishing a coherent and resilient ecological network. Buffering may function by the enhancement or creation of ecotones in the buffer (see 3.1.2.3 **ETPW-109**, **ETPW-117** and **ETPW-213C**), but buffer zones more generally improve the quality and effective area of target sites (Lawton et al. 2010).

3.1.3.2 Co-Benefits and Trade-offs

There will always be a loss of habitat to create buffer zones; presumably this habitat will be of lower quality in general than what is installed in its place, but some ecotonal species could suffer, depending upon the precise habitat change that occurs. Increasing effective habitat patch sizes will also inevitably reduce landscape heterogeneity, with potential negative effects on species that respond positively to mosaic habitats. Conversely, while the buffer habitat’s primary function is to protect the core area, it may also provide various ecological and ecosystem service in its own right.

[TOCB Report-3-6 *Carbon* **ETPW-108**] Any conversion of land cover has the potential to affect carbon sequestration, though a change in above or below ground biomass. The size and the direction of effect is dependent on the specific habitats being converted during the creation of the buffer, the extent of the buffer and creation and management practices. For the effects of creation of specific habitats see section 3.5 in the carbon sequestration review Report-3-6 *Carbon*. Under the assumption that this action involves creating an uncultivated buffer between priority habitats and arable agricultural land, there is likely to be an increase in local above and below ground carbon sequestration rates. If the buffer is against pastoral agriculture the directionality of the effect of buffer creation on carbon sequestration is less clear.

Food and fibre production	Area under production or yield and outside of ELM	**N**
Global, regional & local climate regulation	Above ground carbon sequestration	**T***
	Below ground carbon sequestration	**T***

Duplicated evidence: **EBHE-080**: Create/ manage buffer strips around heritage assets on the shine database that are not Listed Buildings or Scheduled Monuments or around Scheduled Monuments (link boundary features) on cultivated land

3.1.3.3 Magnitude

Specific evidence for effect magnitude does not exist and this will be highly variable between habitat types and contexts.

3.1.3.4 Timescale

Effect timescales will be variable with habitat context, habitat type and the precise threat that is under consideration. For example, the simple imposition of a buffer will create distance between some threats and sensitive habitats instantaneously, while those requiring interception by a given vegetation type will need sufficient time to elapse for the vegetation to mature, i.e. months to a year for herbaceous vegetation and decades for woody vegetation.

3.1.3.5 Spatial Issues

By definition, this management needs to be targeted spatially appropriately with respect to sensitive habitat locations to be effective.

3.1.3.6 Displacement

Some currently non-priority habitat land will inevitably be lost, so production from that land is likely to be displaced elsewhere. Buffer areas may have some intrinsic benefits (see Co-benefits and Trade-Offs), but these are unlikely to be equivalent to the production benefits that have been lost.

3.1.3.7 Maintenance and Longevity

Basic buffer functions are likely to persist inevitably and to require minimal maintenance, given that any benefits from the buffer areas per se are regarded as additional. These additional benefits, however, could depend on the condition of the habitat in the buffer and therefore be dependent upon habitat management for maintenance.

3.1.3.8 Climate Adaptation or Mitigation

As per Lawton et al. (2010), this action should help to promote adaptation to climate change by enhancing the quality of the habitat matrix.

3.1.3.9 Climate Factors / Constraints

N/A

3.1.3.10 Benefits and Trade-offs to Farmer/Land-manager

This will inevitably lead to loss of land that might otherwise be available for production. Larger, more contiguous areas of semi-natural habitat are likely to increase both the ecosystem services and disservices that the semi-natural areas provide to adjacent farmland, e.g. reservoirs of beneficial and problem species, where relevant.

3.1.3.11 Uptake

There may be negative perceptions for landowners based on the issues raised in the section above.

3.1.3.12 Other Notes

None

3.1.4 ETPW-109, ETPW-117 and ETPW-213C: Reduce the hard boundaries between habitats and encourage ecotones / Manage mosaics and natural transitions to other habitats / Create, enhance or manage habitat in new strategic areas to enhance ecological connectivity between existing habitat patches

These three actions describe aspects of habitat management that influence connectivity between patches of semi-natural habitat, although ‘mosaics’ could also refer to different forms of agricultural management, notably approaches to grass mowing, as are used to enhance the quality of intensive grassland for breeding wading birds in the Netherlands. Ecotones are the boundaries between habitat types, here specifically in the sense of habitat zone providing smooth or gradual changes between habitat patches. If such habitats (i.e. vegetation types) soften boundaries between nearby semi-natural patches, such that they are no longer separated by inhospitable land cover that constitutes a barrier (e.g., often, production agriculture), then connectivity is improved. Greater connectivity might also be created by specifically created linking habitat corridors or stepping stones, although Lawton et al. (2010) recommend stepping stones ahead of linear corridors, due to the practicality of securing sufficient land for management. The function of habitat interventions and patch geometries as agents of connectivity will vary between taxa and habitat types.

3.1.4.1 Causality

This is very broad action that could cover a wide range of habitat types from the patch to the landscape scales. Ecotones may be removed as land management hardens the boundaries between habitats, and one approach to habitat creation is to ‘soften’ these boundaries to allow a more gradual transition between habitats or land uses. For example, there are species that require ecotones (the transition between or blending of two different habitats to create (‘messy edges’) for their survival, e.g. Pearce-Higgins et al. 2007), so creating such habitats will benefit these species. Ecotones can be established by creating buffers around priority habitat patches, so this action is related to action **ETPW-108** (section 3.1.2.2), and much of the same considerations regarding evidence apply. Overall, the theoretical action and benefit of this intervention are fairly clear, but the breadth of contexts in which this could apply makes a general statement of specific effects impossible. Evidence of the effects of ecotone creation as a specific intervention seems to be lacking.

The specific case of grassland mosaic management for wading birds, creating patches of long and short grass that are suitable for foraging, nesting, or refuges from mowing or predators, has been well-studied in the Netherlands, where it could be a key measure for meadow grass habitats. The evidence shows some positive responses of target species from trial studies, but also some failures to respond and possible negative impacts (reviewed by the Conservation Evidence project[1]). The implementation of the option in Dutch agri-environment schemes is being monitored, but evidence of its success has yet to be published. Further, it is likely that there are few landscapes in England where this management would be relevant, as there is no obvious analogue for the Dutch meadow grassland context.

[1] https://www.conservationevidence.com/actions/130

3.1.4.2 Co-Benefits and Trade-offs

As with habitat buffers, ecotones will benefit the quality and function of core habitats.

3.1.4.3 Magnitude

Specific evidence for effect magnitude does not exist and this will be highly variable between habitat types and contexts.

3.1.4.4 Timescale

Aspects of ecotonal vegetation will develop within a year and may be functionally complete within a decade, but this will vary depending on the habitat context.

3.1.4.5 Spatial Issues

By definition, this management needs to be targeted spatially appropriately with respect to sensitive habitat locations to be effective.

3.1.4.6 Displacement

Some currently non-priority habitat land will inevitably be lost, so production from that land is likely to be displaced elsewhere.

3.1.4.7 Maintenance and Longevity

Ecotonal habitats may be successional and hence require ongoing maintenance to prevent transition to climax vegetation – wherein natural processes would mean that other land might develop the characteristics of the ecotone, with a greater spatial footprint overall and hence more potential displacement issues.

3.1.4.8 Climate Adaptation or Mitigation

As per Lawton et al. (2010), this action should help to promote adaptation to climate change by enhancing the quality of the habitat matrix.

3.1.4.9 Climate Factors / Constraints

N/A

3.1.4.10 Benefits and Trade-offs to Farmer/Land-manager

This will inevitably lead to loss of land that might otherwise be available for production.

3.1.4.11 Uptake

There may be negative perceptions for landowners based on loss of production land.

3.1.4.12 Other Notes

None.

3.1.5 ETPW-170: Improve management of arable and pastoral farmland to benefit and help secure the long term sustainable recovery of farmland birds, pollinating insects and other wildlife associated with the wider countryside

This action describes a general philosophy or intention, rather than a specific intervention, or set of interventions, whose efficacy can be evaluated. The majority of the English countryside consists of arable and pastoral farmland, so this aim could be achieved by interventions across all of that landscape, or within limited areas (with the goal of an average benefit for the target groups). 'Improving' management in this context, allowing population increases and range expansions in all wildlife that has declined, almost certainly means some reduction in productivity, either from taking land out of production entirely, or by extensifying agricultural systems, i.e. trading yield for resources for wildlife. Systemic changes of this sort have previously been supported by agri-environment schemes, with taxpayer funding allowing farm businesses to continue with the reduction in revenue from production. Therefore, there is a general question of whether such schemes have been effective as a whole, but we do not consider this to be of value here, as schemes are clearly the sum of the actions that constitute their parts and these actions are all reviewed individually in this process. Arguably, a system change to organic farming could be an intervention that fits within this category. Similarly, the general concept of land sparing versus land sharing could be considered here. There is a great deal of evidence relating to these areas; summarising it is beyond the scope of this review and the breadth of activities and interactions involved means that scoring as a single 'action' would not be meaningful, beyond stating the general benefits of improving farmland management for wildlife, which necessarily benefits this 'service' but only benefits others in a limited/context-specific way. The specifics of magnitudes, timings and spatial influences, etc., clearly cannot be identified in a valid way for such a broad suite of changes, so the structure of this review does not support consideration of this broad range of system changes. However, the review also does not clearly allow consideration of combinations of interventions or whole-farm management studies. These general areas are therefore flagged below, but a full review of the benefits or otherwise of each is beyond the scope of this process and would involve a long process for each subject area.

3.1.5.1 Causality

(i) Research considering whole AES agreements, i.e. targeted combinations of interventions at the farm level, has considered only "narrow-and-deep" AES management, i.e. locally intensive management at the farm scale (as opposed to "broad-and-shallow" management spread more thinly across a landscape), either with a single-species or farmland bird-community focus. Management for Cirl Buntings in south-west England has taken an inclusive approach such that the old CS and HLS agreements implemented there have integrated options to provide all the species' requirements, as far as possible. Surveys of breeding and wintering birds to reveal habitat use and population changes have revealed strong increases attributable to the AES management but the management has focused on the conservation outcome, rather than on testing individual option effects and attribution to individual options is difficult (e.g. Peach *et al.* 2001). Thus, the recovery of the species could indicate synergies between different options providing different resources or, more simply, could show effects of the limiting one among the broader suite of options.

A further, medium-term study has examined bird population responses to HLS management in an arable farming area and a mixed farming one, via three-yearly surveys of breeding bird populations at the farm scale. Bright *et al.* (2015) found that Grey Partridge, Lapwing, House Sparrow, Tree Sparrow, Reed Bunting and Yellowhammer increased more on HLS farms than on control farms, while nine other species were non-significant. There was little clear evidence of which AES option types had driven the results because most tests with respect to individual options were non-significant (Bright *et al.* 2015). Walker *et al.* (2018) have conducted a further, improved analysis of the same data, with wider countryside BBS data as a control stratum, thus avoiding potential problems with unbalanced treatment and control samples. There were again local population increases over six years in response to HLS management, although many responses decreased in size over time. This could show a sensitivity to weather events in the effectiveness of the options, but it is also likely that a ceiling will be reached in farm-scale abundance as densities rise locally and this may have occurred in HLS farms with high-quality management. Once again, the significant effects of HLS identified at the farm level were not detectable in terms of the resources provided by individual types of management (which were grouped by types of resource provided for analysis). It is likely that power at the option level was low, especially for the winter food options, because many birds using these options will have bred elsewhere. Likewise, many birds breeding on the study farms probably used habitats outside the farm boundary for at least some of the winter and then possibly responded to the AES management via settlement to breed, rather than through a demographic effect. Thus, the results of this study could represent synergies between multiple option types that are not detectable in tests of individual options, but they do not provide strong evidence to that effect.

(ii) The effects of organic farming have been the subject of numerous studies and reviews. In general, organic management has been found to benefit biodiversity, but it is difficult to apportion benefits to specific aspects of the organic system. For example, organic farms are typically mixed and have field boundary features that provide a livestock control function, so land-use heterogeneity and boundary structure (hedgerow) effects are confounded with those of organic in-field management. Hence, specific negative effects of practices such as mechanical weed control may be obscured and the general, system-level positive effects may not be transferable to the field scale.

(iii) Land-sparing is the spatial separation of environmental management from agricultural production, as opposed to land-sharing, which involves environmentally sympathetic agricultural management, typically involving a reduction in yield. However, there are grey areas between these concepts because of spatial scale: land can be 'spared' at the field scale (e.g. semi-natural boundary vegetation, field margins), as well as in terms of whole landscapes. One strand of modelling studies has predicted clear benefits from landscape-scale land-sparing for biodiversity, due to the greater support for biodiversity from semi-natural, relative to agricultural, habitats (Finch et al. 2019, Lamb et al. 2019). However, these studies assume that spared farmland provides all the ecological functions of established semi-natural habitats, such as mature woodland, and that the remaining production farmland is only intensified sustainably.
Further, the many species that are the subject of conservation concern in farmland as a result of past farmland intensification are likely to be affected negatively, even if biodiversity overall increases due to greater areas of semi-natural habitat being created. The models also do not consider the spatial distribution of spared and production land, which would be likely to lead to polarisation in the landscape, with regions being largely spared or used for production, with unknown consequences for resource protection and landscape connectivity. Overall, a sparing approach would be likely to have strongly positive effects for a range of species, given particular outcomes with respect to spatial distribution, but also probably various negative consequences, and a combination of sharing and sparing, or a range of scales of sparing in different locations, would be most positive overall.

3.1.5.2 Co-Benefits and Trade-offs

The action group is too broad to assess this.

3.1.5.3 Magnitude

The action group is too broad to assess this.

3.1.5.4 Timescale

The action group is too broad to assess this.

3.1.5.5 Spatial Issues

Whether landscapes and be spared in practice, given economic drivers, and the suitability for organic farming, for example, will vary considerably between regions, affecting where such system changes are feasible in practice.

The effectiveness of management in general has been investigated with respect to landscape context. Tscharntke et al. (2005) refer to landscapes with less than 2% semi-natural habitats as "cleared" landscapes, where the effectiveness of conservation is limited by the basic absence of species sources. Landscapes with 2-20% semi-natural habitat in the matrix are referred to as "structurally simple" landscapes, where species sources are still present and conservation initiatives can achieve good results. In "complex" landscapes with more than 20% semi-natural habitats, the productive area is continually colonised by species from the surrounding species-rich landscape. Environmental interventions to benefit biodiversity are most effective in intermediately complex landscapes: impoverished landscapes lack the source populations to allow positive response to occur and rich landscapes are already effectively saturated (Tscharntke et al. 2005).

Evidence from experimental and analytical studies of English farm landscapes shows that bird species vary in the spatial scale at which they show most sensitivity to landscape structure, but that many are most influenced by spatial extents that are much larger than that of most individual land-holdings (Pickett and Siriwardena 2011), and that home ranges similarly commonly extend beyond farm boundaries, even within a season (Siriwardena et al. 2006). This suggests that effective management needs to be coordinated at the landscape scale, or certainly across multiple, neighbouring farms. Tscharntke et al. (2012) consider multiple approaches to investigating landscape influences on local biodiversity responses. Across multiple taxonomic groups, the evidence shows that habitat heterogeneity, which can be delivered by small-scale agriculture or crop diversity at the landscape scale, in different contexts, is able to drive positive biodiversity responses (Fahrig et al. 2011, Sirami et al. 2019, Hass et al. 2018), suggesting a role for the spatial organisation of agriculture in promoting farmland biodiversity.

3.1.5.6 Displacement

The action group is too broad to assess this in detail, but any reduction in yield, alone, is likely to lead to some displacement of production.

3.1.5.7 Maintenance and Longevity

The action group is too broad to assess this; in general, farming systems require ongoing management and support payments are likely to be needed indefinitely.

3.1.5.8 Climate Adaptation or Mitigation

The action group is too broad to assess this.

3.1.5.9 Climate Factors / Constraints

The action group is too broad to assess this.

3.1.5.10 Benefits and Trade-offs to Farmer/Land-manager

The action group is too broad to assess this, but system changes in general will generally require a change of business model, which will attract some and deter others.

3.1.5.11 Uptake

The action group is too broad to assess this, but system changes in general will generally require a change of business model, which will attract some and deter others.

3.1.5.12 Other Notes

None.

3.1.6 ETPW-163: Support farmers and land managers to undertake translocations where these have clear benefits for biodiversity, ecosystem services and landscape transformation, including appropriate land management, and facilitation of these … / ETPW-164: Protect new colonists of conservation value

We assume that this action category refers to species translocations. These could aim to facilitate system adaptation to climate change, i.e. introducing new species or varieties from elsewhere in the range with warmer climates, aim to deliver compensatory movements in the face of developments (newts or reptiles, effectively bypassing natural colonisation), or comprise re-introductions of previous extirpated species (e.g. red kites, white-tailed eagle, beaver, curlew). **ETPW-163** would involve artificial introductions and **ETPW-164** the protection of natural colonisations, but with probably similar actions in practice after the initial events.

3.1.6.1 Causality

Thomas (2011) argues that translocating threatened species to locations where their survival under climate change would be more likely is the only way to preserve such species and that the risks of so doing are low if the destination areas are chosen with care. It remains the case, however, that outcomes cannot be predicted with certainty and that failures to establish have been common, although lessons from these failures can improve future efforts (Berger-Tal et al. 2020). With conservation-based guidance, unintended negative consequences of translocations are very rare (Novak et al. 2021). Well-planned translocations therefore have the potential to be valuable, although they do not always succeed, and effects will vary with context and on a case-by-case basis. A review has indicated that as little as 25% of translocations succeed and that, even then, they may not deliver desirable metapopulation or ecosystem outcomes (Bradley et al. 2022).

Great-crested newts *Triturus cristatus* represent probably the most commonly translocated species in the UK, with breeding colonies frequently being moved to new ponds during developments. Habitat recreation has been partial on average but, although follow-up monitoring has not always occurred, there was evidence of breeding at most sites one-year post-development (Edgar et al. 2005). Nevertheless, it is unclear whether the translocated populations were sustainable in the long-term (Edgar et al. 2005). Within overall-successful translocations, some receptor sites perform better than others, so success is not a given and monitoring is essential (Harper et al. 2017, 2018). In general, the Conservation Evidence project found that amphibian translocation was mostly beneficial (Natterjack Toad being a further key species in the UK), with little evidence of harms, but would not always succeed (Smith et al. 2018).

Reintroductions of extirpated species cannot be generalised as the conditions, processes, actions and monitoring will be as diverse as the species themselves. However, there are clear success stories, such as the Red Kite *Milvus milvus* in Southern England (Carter & Newbery 2004) and short-term stability of small European Beaver *Castor fiber* populations across England (Halley et al. 2021). Conversely, the reintroduced Corncrake *Crex crex* population in Cambridgeshire was being sustained by ongoing releases and long-term sustainably is unknown (Wotton et al. 2015).

3.1.6.2 Co-Benefits and Trade-offs

These will be minimal, as most species involved will be uncommon and will have little ecosystem impact. However, effects of some species could be large and every species and context ought to be evaluated individually. New ponds may be needed for newts, as an example of habitat creation that is likely to benefit some other species, but to harm others. Beavers are ecosystem engineers, potentially having large impacts on local habitats and flooding regimes, again with both positive and negative impacts on different features of the environment.

3.1.6.3 Magnitude

This will be hugely variable between species and contexts.

3.1.6.4 Timescale

This will be hugely variable between species and contexts.

3.1.6.5 Spatial Issues

Accurate, sound scoping of introduction sites or natural sites subsequently to protect is likely to be critical for success, so spatial location will be key, but ‘good’ locations are not predictable in general terms.

3.1.6.6 Displacement

This will mostly be low, but some habitat changes will be needed in some case, leading to likely displacement of production.

3.1.6.7 Maintenance and Longevity

Long-term, sustained introductions may be necessary, rather than single events, and ongoing habitat management or site protection is likely to be beneficial, if not essential, until populations are self-sustaining. Monitoring will be critical to inform these actions.

3.1.6.8 Climate Adaptation or Mitigation

Some reintroductions are concerned with climate adaptation, by definition; others could readily be adapted to do so by choice of site and the management that is undertaken there.

3.1.6.9 Climate Factors / Constraints

Success may well be climate-dependent and sensitive to climate change, so future climate should be taken into account in choosing and managing sites for reintroduction and protection, as it is likely often to affect long-term viability.

3.1.6.10 Benefits and Trade-offs to Farmer/Land-manager

There are likely to be few benefits to land managers, but requirements for species or habitat management may restrict other activities. However, all cases will be different, and flood management benefits from beaver activity, for example, might be significant.

3.1.6.11 Uptake

Many species are neutral, but some are controversial and farmer/public attitudes may be significant in preventing or facilitating the initiation or success of reintroductions.

3.1.6.12 Other Notes

None

3.1.7 ECCA-047: Source trees from biosecure suppliers

When new woodland is created or trees are planted for other habitat interventions, saplings need to be sourced from commercial suppliers. These could be based on the UK or in continental Europe, but could entail the introduction or spread of disease and/or invasive organisms, for example within potting compost or within or upon the vegetation of the young trees. These unintended negative consequences should clearly be avoided, which sourcing from biosecure suppliers should prevent.

3.1.7.1 Causality

This action is essentially preventative, aiming to avoid unintended problems. Evidence for its success is therefore impossible to find and the support for introducing such a measure comes from evidence for the effects of not doing it (e.g. Lilja et al. 2011) or assessment of potential problems from sampling of commercial products, e.g. the presence of plant pathogens and fungi on commercial seed (Cleary et al. 2019).

3.1.7.2 Co-Benefits and Trade-offs

[TOCB Report-3-6 carbon **ECCA-047**] There is no empirical evidence for the effect of sourcing trees from a biosecure supplier on carbon stocks and sequestration directly, however this action is supported by a robust logic chain, given that tree disease and pests restrict carbon sequestration via tree morbidity and mortality

(Hill et al., 2019; Quirion et al., 2021). Expert judgement suggests that the effectiveness of using biosecure suppliers will depend on the likelihood of disease of pest invasion through other avenues, although delaying or reducing the scale invasive species establishments can increase the opportunity for management and enable more effective responses to emerging issues.

Food and fibre production	Area under production or yield and outside of ELM	**N**
Global, regional & local climate regulation	Above ground carbon sequestration	**LT****
	Below ground carbon sequestration	**LT***

3.1.7.3 Magnitude

This depends entirely on the potential severity of avoided threats, which will be highly variable and case-specific.

3.1.7.4 Timescale

N/A

3.1.7.5 Spatial Issues

None.

3.1.7.6 Displacement

N/A.

3.1.7.7 Maintenance and Longevity

New threats are likely to arise as the climate changes, both with new diseases and new source markets, plus existing sources being unable to persist. This will require ongoing renewal of supplier lists and problem organisms. Ongoing activity will be required while trees are being sourced.

3.1.7.8 Climate Adaptation or Mitigation

N/A

3.1.7.9 Climate Factors / Constraints

New threats are likely to arise as the climate changes, both with new diseases and new source markets, plus existing sources being unable to persist. This will require ongoing renewal of supplier lists and problem organisms.

3.1.7.10 Benefits and Trade-offs to Farmer/Land-manager

Avoided imported threats may be a latent, cryptic benefit, but near-impossible to measure.

3.1.7.11 Uptake

Product pricing could be an issue, if biosecure sources are more expensive than insecure suppliers.

3.1.7.12 Other Notes

None.

3.1.8 ECPW-247: Use precision systems such as spot spraying and weed wiping when applying a pesticide

Pesticide applications – whether aimed at invertebrates, plants or fungi – can have negative effects on non-target species as well. In addition, pesticide applications to parts of fields where they are not needed because the pest is not present are clearly inefficient. Hence, precision applications targeted at problem areas should have both biodiversity and economic benefits, once surveys of the affected fields have identified where the need is greatest.

3.1.8.1 Causality

There is good evidence for negative effects of pesticides, particularly indirect effects, on non-target and beneficial invertebrates, in respect of modern types of chemical input. This suggests that reducing inputs will be beneficial, as non-target species will then be able to recover. However, the extent of this recovery will depend on seed banks and source populations, so is not a given. Pesticide effects on higher trophic levels are less clear, probably because the abundance of the invertebrates that are affected is not a critical, limiting factor for birds or mammals. There is, therefore, rather equivocal evidence for the need and therefore potential benefit of this action, but reductions in inputs in general can only be positive for the farmed environment.

In terms of efficacy, the design of effective weed-wiping technology is still in development, with further research needed to optimise systems for weed control and the absence of effects on non-target species (Harrington & Ghanizadeh 2017). In combination with lenient grazing, herbicide wiping for the control of *Cirsium arvense* in grasslands achieves sufficient control and hence removes the need for broad herbicide application, but with a small decrease in non-target plant occurrence, and lenient grazing alone can achieve the same effects; further, longer-term trials are recommended (Pywell et al. 2010). Similarly, automated spot-spraying delivered lower impacts on non-target plants, but also less impact on target weeds, in one study (Power et al. 2013). In general, there is limited evidence for the biodiversity benefits of this specific action and further research seems widely to be called for, but there are strong hypothetical reasons why it should be beneficial.

3.1.8.2 Co-Benefits and Trade-offs

Agronomy and production efficiency should be increased with lower input costs, but precision-targeting could be more labour-intensive or require expensive equipment.

3.1.8.3 Magnitude

Locally high magnitude effects may occur, i.e. within fields, but long-term application is likely to be required to deliver major benefits.

3.1.8.4 Timescale

Pesticide benefits for production are seen within a cropping season, but long-term (multi-year) application is likely to be required to deliver major benefits, i.e. on seed-banks or invertebrate populations.

3.1.8.5 Spatial Issues

All benefits of actions reducing herbicide inputs are dependent upon the seed bank or colonisation of conservation-target species, as these species cannot respond unless they are present. With many seed banks depleted by decades of herbicide use and farmers potentially targeting fields for reduced-input cropping where weed burdens are lower, there may be strong spatial constraints on efficacy.

3.1.8.6 Displacement

None

3.1.8.7 Maintenance and Longevity

To be effective, this needs to be applied indefinitely, in every cropping season.

3.1.8.8 Climate Adaptation or Mitigation

N/A

3.1.8.9 Climate Factors / Constraints

N/A

3.1.8.10 Benefits and Trade-offs to Farmer/Land-manager

There may be significant equipment costs, but benefits from reduced quantities of chemicals applied. The balance and net benefit will depend on ELMS payment rates.

3.1.8.11 Uptake

As well as cost issues, farmers may be unwilling to change long-established practices, i.e. there could be an element of behavioural inertia.

3.1.8.12 Other Notes

None

3.1.9 ECPW-273: Establish the resistance status of pests on farm

Pesticides can have negative effects on a wide range of non-target species as well as target pests. Moreover, long-term selection pressure on pests often results in the evolution of resistance to pesticides, making them less effective. It should, therefore, reduce unnecessary negative effects on non-target species if applications of pesticides where a priori resistance among pests is high are avoided. Economic efficiency would also be enhanced. This requires the identification of the resistance status of the relevant pests before a best pesticide solution to the local problem is finalised.

3.1.9.1 Causality

Alone, knowing resistance status would not confer any environmental benefit, but coupled with informed choice and use of chemical inputs or other control strategies it should reduce overall input levels and therefore deleterious effects on non-target species (see. 3.1.3.2 **ECPW-247**: Use precision systems such as spot spraying and weed wiping when applying a pesticide). There appears to be no direct evidence regarding the environmental effects of this action and it is entirely dependent upon appropriate follow-up actions being taken.

3.1.9.2 Co-Benefits and Trade-offs

There should be agronomic and production efficiency benefits from reduced requirements for inputs.

3.1.9.3 Magnitude

Effects may be small in practice because inputs are not changed much, but this depends on the pest threat and the nature and quantity of the inputs that are avoided.

3.1.9.4 Timescale

Pesticide benefits for production are seen within a cropping season, but long-term (multi-year) application is likely to be required to deliver major benefits, i.e. on seed-banks or invertebrate populations.

3.1.9.5 Spatial Issues

All benefits of actions reducing herbicide inputs are dependent upon the seed bank or colonisation of conservation-target species, as these species cannot respond unless they are present. With many seed banks depleted by decades of herbicide use and farmers potentially targeting fields for reduced-input cropping where weed burdens are lower, there may be strong spatial constraints on efficacy.

3.1.9.6 Displacement

None

3.1.9.7 Maintenance and Longevity

To be effective, this needs to be applied indefinitely, in every cropping season.

3.1.9.7.1 Climate Adaptation or Mitigation

N/A.

3.1.9.8 Climate Factors / Constraints

N/A

3.1.9.9 Benefits and Trade-offs to Farmer/Land-manager

There should be agronomic and production efficiency benefits from reduced requirements for inputs.

3.1.9.10 Uptake

No issues.

3.1.9.11 Other Notes

None

3.1.10 ETPW-001: Use a biosecurity plan

Bringing any organic or biological material onto a farm, including inadvertent transport via clothing, vehicles, livestock or wild animals, has the potential to entail the introduction or spread of disease and/or invasive organisms. This could have unintended negative consequences for agriculture or biodiversity and should clearly be avoided, which a clear plan for biosecurity procedures should prevent as far as possible.

3.1.10.1 Causality

Alone, a biosecurity plan would not confer any environmental benefit, but following a well-designed plan would prevent problems occurring. As in **ECCA-047** (section 3.1.3.1), a plan is essentially preventative, aiming to avoid unintended problems. Evidence for its success is therefore impossible to find and the support for introducing such a measure comes from the potential risks of not doing it. There appears to be no direct evidence regarding the environmental effects of this action and it is entirely dependent upon appropriate actions being taken, but the logical basis is clear.

3.1.10.2 Co-Benefits and Trade-offs

There could be agronomic benefits from a reduced need for inputs.

[TOCB Report-3-6 *Carbon* **ETPW-001**] There is no empirical evidence for the effect of sourcing trees from a biosecure supplier on carbon stocks and sequestration directly, however this action is supported by a robust logic chain, given that tree disease and pests restrict carbon sequestration via tree morbidity and mortality (Hill et al., 2019; Quirion et al., 2021).

Food and fibre production	Area under production or yield and outside of ELM	**N**
Global, regional & local climate regulation	Above ground carbon sequestration	**T**
	Below ground carbon sequestration	**T**

3.1.10.3 Magnitude

This is unpredictable and would be highly variable between different potential biosecurity threats.

3.1.10.4 Timescale

This is unpredictable and would be highly variable between different potential biosecurity threats.

3.1.10.5 Spatial Issues

The benefits from avoiding biosecurity threats will be greatest where the threats are most active, which is likely to vary geographically with sources of disease, etc.. Hence, spatial targeting will influence the size of the potential benefit, but there will be no disbenefit in using a plan where risks are lower.

3.1.10.6 Displacement

None.

3.1.10.7 Maintenance and Longevity

Plans would need to be long-term, ongoing or regularly renewed, with management to adhere to them being indefinite.

3.1.10.8 Climate Adaptation or Mitigation

N/A

3.1.10.9 Climate Factors / Constraints

Biosecurity threats may change with climate change, so the value of plans could change and may well increase as a means of prevention of unforeseen future problems (although this is impossible to measure).

3.1.10.10 Benefits and Trade-offs to Farmer/Land-manager

There could be agronomic benefits from a reduced need for inputs.

3.1.10.11 Uptake

Costs of biosecure management may be higher than those of non-biosecure activities, and hence prove to be a barrier to the implementation of a plan.

3.1.10.12 Other Notes

None

3.1.11 ETPW-008: Monitor and control damaging terrestrial animal species (e.g. deer, grey squirrel)

Some animal species, particularly certain invasive non-native species but also some nominally native or long-established ones, could have detrimental effects on other biodiversity (effects on agricultural production are out of scope for this review). Therefore, monitoring local occurrence, abundance and effect of these species could be useful to identify where problems are occurring, and subsequently inform where control measures might be valuable. The latter measures may then have biodiversity benefits. It is not clear which species other than deer and grey squirrel may be in scope for this action, and there is considerable subjectivity and context-dependence in whether species are considered to be 'damaging', so the review below is focused only on the aforementioned taxa, together with rabbit, which is singled out in a separate action (**ETPW-152** Manage damaging rabbit populations). It is also not clear what types of ecological/environmental 'damage' are in scope: for deer, a key issue is woodland vegetation structure and the various species that depend upon it; for grey squirrel, key issues are competition with red squirrel and predation of birds' nests, as well as damage to trees (Mayle & Broome 2013).

Other species that have been implicated as damaging to conservation-target species in terrestrial habitats include probably all extant predatory mammals and a range of predatory bird species. Evidence relating to these species and their 'control' is considered in section 3.1.4.2 (**ETPW-212**).

There are two levels to evaluating the effectiveness of this intervention: first, the extent of the problem caused by the putatively damaging species and, second, the efficacy of the control measures employed. The accuracy and reliability of practicable monitoring approaches may also be critical. Note that the specific interventions that could be applied to the target taxa are very different, so any assessment of efficacy, etc., will be inexact.

3.1.11.1 Causality

Deer have been identified as severe effects on the vegetation structure in woodland, with knock-on effects on bird communities in particular (Fuller & Gill 2001, Holt et al. 2011, 2014, Newson et al. 2012, Eichhorn et al. 2017). This effect involves invasives such as Reeves' Muntjac, but also long-established introductions such as Fallow and Sika Deer, and expanding native species in Red and Roe Deer. Interventions could involve shooting or fencing. Shooting with the aim of population reduction would need to be coordinated over a large area to account for the large home ranges of deer. As with other animal control measures (see **ECPW-251**), there is a lack of direct evidence for the effectiveness of different approaches and intensities (levels of effort) in control or culling of deer, such as would be needed to design a cost-effective control programme. For example, Ueno et al. (2010) used modelling from sex-specific harvest data to predict the number of hunter-days that would be required to prevent population increases in a deer population in Japan, finding that population growth rate changed more in response to annual changes in recruitment rate than hunting mortality rate. In the UK, there have been few relevant studies, but some models of harvest intensity provide some relevant information (e.g. Trenkel 2001) and it is known that the effects of a given landscape-level density of deer can be highly dependent upon habitat context (Spake et al. 2020).

Deer fencing can practically only be conducted at the plot level, i.e. enclosing individual woodland stands, but is effective at changing habitat structure and dependent bird populations, at least (Holt et al. 2014).

Grey Squirrel is an introduced predator of nesting birds, but may mostly fill the niche of the native Red Squirrel, so may have little additive effect (Newson et al. 2009). If this impact is of interest, evaluations of effects of squirrel removal therefore also need to account for what might replace them, ecologically. Interventions relevant to squirrel impacts could involve preventative measures such as nest protection, as well as lethal removal (trapping or shooting) and immunocontraception (Barr et al. 2002), but only control measures seem to be in scope for the option as defined.

Perhaps the major conservation priority for grey squirrel control is its effect on the remaining red squirrels and allowing the latter populations to persist and to expand. Trapping grey squirrels can both reduce densities and reduce the prevalence of the virus that mediates negative competition effects on red squirrels (Travaglia et al. 2020), but modelling analyses suggest that both ongoing grey squirrel removal and recovery of a key predator, pine marten *Martes martes*, would be required to allow red squirrel persistence (Twining et al. 2021). Grey squirrel control alone would require intensive, long-term and continuous effort to deliver a sustained population effect in practice, but pine marten recovery combined with control of grey squirrels in urban refugia has the potential to allow red squirrel recovery (Twining et al. 2021).

Rabbits are not clearly identified as being damaging to other biodiversity, but are strong ecosystem engineers in various semi-natural habitats, so influence habitat structure and suitability for other taxa in contexts such as downland and heathland, in some cases being critical for habitat maintenance (e.g. Dolman & Sutherland 1992, Crawley 1990, Williamson 1975, Green & Taylor 1995, Thomas et al. 1992).

3.1.11.2 Co-Benefits and Trade-offs

Some 'problem species' may be problems for production as well as biodiversity, or to the game industry, so removal or reduction could have economic benefits.

[TOCB Report-3-6 *Carbon* **ETPW-008**] Terrestrial herbivores can cause significant damage to vegetation as a result of overabundance, with consequences for carbon stocks and sequestration rates and the mechanisms for this are well established (Côté et al., 2004; Hirst, 2021; White, 2012). Herbivory pressure can affect carbon sequestration in the short term through the removal of living biomass and by triggering the reallocation of sequestered carbon away from sinks associated with long term storage.

Evidence suggests that shaded seedlings are particularly vulnerable to browsing pressure by deer (Harmer & Gill, 2000). Browsing and seed or nut predation can reduce long term carbon sequestration by inhibiting vegetation regeneration, either in planned natural regeneration efforts or by causing recruitment failure existing woodland (Turczański et al., 2021; White, 2012; Willoughby et al., 2011). The impacts of damaging terrestrial species will species vary with foraging strategy and browsing preferences (Harmer & Gill, 2000), and different management options will be available as a result, making the monitoring or the presence of damage caused by terrestrial herbivores critical to effective management.

A review of the impact of deer browsing on carbon sequestration in Scottish woodlands found that there was limited evidence quantifying the effects of deer browsing on carbon sequestration in the short and long term, but estimated that reducing deer densities to below 5 deer km^{-2} would facilitate natural regeneration of Scottish woodland (Hirst, 2021). Putman, (2011) reported that unfenced native woodlands could regenerate naturally if there are fewer than 4–5 large deer or fewer than 25 roe deer per 100ha, but that deer density alone was unlikely to be a good predictor of impact. Low level browsing in mature woodland is likely to have a much smaller effect on carbon sequestration and may play an important role in nutrient cycling. Squirrels are also known to cause damage to trees via bark stripping, evidence for which is wide spread, particularly in the South East of England (National Forest Inventory, 2020).

Bark stripping is thought to increase susceptibility to infection and reduce timber, and has been shown to increase the probability that a tree will lose its leading shoot (Mayle et al., 2009), but there is no estimate of the impact of this on carbon sequestration nationally.

sensitive species in a larger, more connected problem species' population.

3.1.11.3 Magnitude

Not assessed

3.1.11.4 Timescale

Not assessed

3.1.11.5 Spatial Issues

Not assessed

3.1.11.6 Displacement

None

3.1.11.7 Maintenance and Longevity

Grey Squirrel populations recover from control measures quickly (Lawton & Rochford 2007), so continuous control efforts are likely to be required. As above, this is likely to be the case especially in larger, more connected populations of problem species, in particular.

3.1.11.8 Climate Adaptation or Mitigation

N/A

3.1.11.9 Climate Factors / Constraints

N/A

3.1.11.10 Benefits and Trade-offs to Farmer/Land-manager

Some 'problem species' may be problems for production as well as biodiversity, or to the game industry, so removal or reduction could have economic benefits.

3.1.11.11 Uptake

Control measures can be unpopular with the public, even when invasive species are involved, which could impact on efficacy and impact of management. For example, coordinated measures at the landscape scale could be compromised if some landowners do not engage and hence provide a refuge for the population of the problem species. However, this issue is being addressed actively in at least one ongoing project aiming to facilitate deer management by enhancing knowledge exchange with land managers about the balances between the negative effects of deer, and their cultural and economic value, via aweb-based tool ('iDeer', based at Reading University; https://www.uktreescapes.org/projects/ideer/).

3.1.11.12 Other Notes

None

3.1.12 ETPW-247: Take weed/pest/ disease infested areas out of cropping and convert to a suitable alternative

Some disease, weed and animal pest diseases can persist in the soil in affected locations and therefore be difficult to eradicate, hence requiring annual treatment of 'symptoms', as opposed to definitive, single actions. A solution to this is to remove the areas concerned from agriculture (generally cropping) and convert them to another use, either temporarily or permanently, depending on the persistence of propagules in the soil. The primary benefit of this intervention would be agricultural/agronomic, but the reduction in chemical inputs and habitat change (probably increasing on-farm heterogeneity) could also benefit biodiversity.

3.1.12.1 Causality

This is an action for agricultural rather than biodiversity benefit, but conversion of intensive farmland into other habitat has the potential to benefit various species and wider communities depending upon what the alternative habitat is: semi-natural alternatives are likely to provide clear biodiversity increases and benefits at a landscape scale from increased habitat heterogeneity, but change to another agricultural use is likely to be more equivocal. Regardless, a specific assessment cannot be made without detail of replacement land-use being known, and effects will vary according to what the alternative is and where it is. The evidence here can only involve a logic chain, as the subject is too broad for generally relevant studies to have been conducted. Evidence for specific habitat change effects would a very large potential set of study contexts, review of which is not practicable in the time available here.

3.1.12.2 Co-Benefits and Trade-offs

This action is primarily for agronomic benefit and other effects are hard to predict, especially the breadth of the action category.

3.1.12.3 Magnitude

The action category is too broad to allow assessment of this, as effects are too unpredictable.

3.1.12.4 Timescale

The action category is too broad to allow assessment of this, as effects are too unpredictable.

3.1.12.5 Spatial Issues

The action category is too broad to allow assessment of this, as effects are too unpredictable.

3.1.12.6 Displacement

Displacement of the original land-use is very likely to occur if the overall farming system and production regime is to be maintained, so any creation of habitat may be offset with losses elsewhere.

3.1.12.7 Maintenance and Longevity

The action category is too broad to allow assessment of this, as effects are too unpredictable: it depends on what the replacement land-use involves.

3.1.12.8 Climate Adaptation or Mitigation

Depending on the replacement habitat, there could be climate mitigation benefits from habitat connectivity, etc...

3.1.12.9 Climate Factors / Constraints

None.

3.1.12.10 Benefits and Trade-offs to Farmer/Land-manager

Any change in farming system or land-use will have effects on farm economics or the business model, which could be a significant trade-off against profitability.

3.1.12.11 Uptake

Any change in farming system or land-use will have effects on farm economics or the business model. This could clearly prove to be a barrier for uptake by landowners.

3.1.12.12 Other Notes

None.

3.1.13 ECPW-251: Monitor and record natural predator levels and ETPW-212: Manage predation sustainably

We assume that these actions relate to vertebrate predators of other vertebrates; predation of and by invertebrates is considered in respect of biological control of pests (see **ECPW-269**), and we are unaware of any research or policy interest in predation as an influence on the conservation of invertebrates. The outcome to benefit would be biodiversity in the form of priority species, potentially also the resilience of populations to climate change effects, if the additional stressor of predation impacts is reduced.

Predation is a natural ecological process and a wide range of specialist and generalist predators are integral to ecosystems. However, predation obviously has an impact at least on the individuals that are predated and the abundance/density of predator species varies in space and time. Hence, it is possible that negative impacts of predation (as a direct cause of mortality, or an indirect cause, via disturbance) are critical for conservation target species. The 'control', i.e. killing and removal, with the aim of minimising local abundance, of predators has long been a feature of the management of populations of both introduced (e.g. Pheasant *Phasianus colchicus* and Red-legged Partridge *Alectoris rufa)* and native (e.g. Red Grouse *Lagopus lagopus* and Grey Partridge *Perdix perdix*) game species. The ultimate aim of this management is to maximise quarry species' numbers for the autumn shooting season, so shooting interests are effectively in competition with natural predators, for whom the annual production of young birds is a food source supporting their own reproduction. This has led to the development of a wide range of 'control' techniques, involving trapping, shooting and poisoning. The application of many of these measures to taxa such as crows and foxes remains legal (as exceptions within the Wildlife and Countryside Act 1981), although illegal removal of still-protected species is still believed to be common (e.g. Smart et al. 2010), and the 'control' of otherwise protected species such as the Common Buzzard *Buteo buteo* has also been licensed in specific circumstances (Natural England 2016). A combination of a perception that predation has been an important negative influence on conservation target species and studies showing correlations between target bird species abundance or demography and predator removal measures at the site scale has led to proposals for the latter to be used in large-scale conservation management.

The actions as titled do not capture what would actually be involved in predator management very clearly. Monitoring natural predator levels (**ECPW-251**) can never have an effect alone and also needs reference levels in space or time to allow interpretation, i.e. are levels 'low' or 'high', presumably to be followed by action to alter them (a different action). Whether the subsequent management, if successful, would then have any impact would be dependent upon whether predation genuinely had a population effect. Managing predation sustainably (**ETPW-212**) is an aspiration rather than an action – 'predation' is a process that is not simply related to predator type or density. Therefore, this potentially encompasses measures such as fencing of sensitive habitats, diversionary feeding and the removal and hand-rearing of vulnerable eggs or nestlings, as well as measures to manipulate predator densities, and involving predators that currently can and cannot be killed legally. Further, 'sustainability' in this context could refer to the activities that are undertaken themselves, so the human and physical infrastructure to support predator removal or nest protection in the long term, for example, or to the ecological processes, i.e. the maintenance of viable native predator populations under the pressure of removal activity or the maintenance of a level of predation impact that is deemed not to be damaging to populations of prey species. Whatever the specific definition of interest, there is a wide range of individual interventions that could be employed under this 'action' including habitat management, culling of predators, employment of gamekeepers, nest protection, fencing, diversionary feeding, etc..

The evidence for and efficacy of all elements of predator management could be critical to decision-making over the implementation of specific intervention.

3.1.13.1 Causality

Given the range of possible interventions here, it is not practical to conduct a comprehensive review without greater specificity and it could be damaging to make recommendations regarding the overall broad aspiration that could then be interpreted as support for specific interventions within the set. These specifics are likely to vary hugely in their efficacy, the strength of the relevant evidence base. Further, predator 'control' is usually conducted in combination with other management activities to benefit game species, making its specific effect impossible to discern (e.g. White et al. 2008, Mustin et al. 2018). Some key issues regarding the relevant evidence are described below.

Smith et al. (2010) conducted a systematic, international review of predator removal, concluding that removing predators increased hatching success, fledging success and breeding populations, and that removing all predator species achieved a significantly larger increase in breeding population than removing only a subset, but also identifying various ethical and practical problems. There are also important contrasts between management of closed populations on islands (where much predator removal research has occurred) and mainland populations (where permanent predator removal is not feasible). Côté & Sutherland's (1997) review found that predator removal increased breeding success and post-breeding population sizes (i.e. the target parameter for game management), but had widely varying effects on breeding populations. This is likely to reflect variation in the importance of predation by the species that are removed as a limiting factor for prey populations, as well as behavioural effects such as prey aggregation in areas of low predator density that do not reflect overall population increases. In the right circumstances, i.e. where there are negative effects of particular predators on populations of target species, at least local and short-term benefits of predator removal can be achieved. However, the ecology of these situations may be complex, with multiple potential predators being present, competing with one another and replacing one another's impacts. The badger cull in south-west England involved a well-controlled trap-and-shoot method and successfully reduced badger densities, but has had no demonstrable impact on bird population growth rates or diversity, either because reduced predation impacts were replaced by pressure from other predators, or because predation was not a significant factor in population growth (Kettel et al. 2020, Ward et al. 2022). At the landscape scale, considering national populations, studies in the UK have failed to find evidence for negative effects of generalist predatory birds on passerine population change (Thomson et al. 1998, Newson et al 2010). Further, successful predator removal is itself an outcome, not an action: in a mainland context, removal efforts need to be continual and can even lead to increases in predator abundance (Calladine et al. 2014). For example, the effectiveness of predator removal measures on lapwing breeding success has been found to be variable between sites and also to have no overall effect on population trends (Bolton et al. 2007). White et al. (2014) found that predator reduction, combined with habitat management, had a positive effect on five out of six species at the farm scale, but not all, and that only one species showed a positive effect on a farm with an intermediate level of control effort, with one species there actually being negatively affected.

It is also generally unknown what level of residual predator abundance would lead to acceptable prey numbers; note that game management operates in practice to minimise numbers of perceived problem species, not to maintain them at a sustainable level. The consequences of hypothetical, landscape-scale, coordinated removal efforts on generalist predators are not known: metapopulation effects might maintain numbers at a low level from (re)colonisation from habitats like urban areas in which there is no control activity, or populations could crash and become a conservation problem themselves. Overall, measures aiming to effect lethal removal of predators are likely to be effective in the conservation of target species in some circumstances, but will also be a long-term, expensive commitment, and the individual, actual, removal actions need to be evaluated for possible efficacy, including the responses of other predators in the ecosystem, not judged on the concept or outcome of particular, resulting levels of predator abundance (see, e.g., Bolton et al. 2007). For a targeted AES intervention, more specifics are needed than 'control predators':

these are native species and clear guidance as to specific actions (e.g. trapping, shooting) or numbers or proportions to remove, or control effort to be expended (e.g. trap-days), are needed to produce coherent actions whose potential impact can then be assessed critically.

There is little evidence for the effects of illegal removal of predators on prey because of the obvious problems with data recording, although Mustin et al. (2018) reported that seven studies found only negative effects on other species. Note that there is a likely bias in the literature in this area in that studies of illegal control are probably predisposed to be seeking negative effects, just as those of legal control are largely supported by game industry concerns with a vested interest in finding positive results.

Smith et al. (2011) conducted a systematic review of the effects of methods of excluding nest predators from access to vulnerable nesting birds, finding that predator exclusion using either exclusion fences or nest-cages resulted in a significant increase in hatching success, but that cages (which are also labour-intensive) can also have negative effects on adult survival and that overall effects on populations are unknown. The latter will critically depend on whether hatching success is limiting for abundance, which will be case-specific. Electric fencing can improve lapwing breeding productivity by excluding large mammalian predators, with effectiveness varying with different types of fence (Rickenbach et al 2011, Malpas et al.2013), so the details of the measure applied here are important

When predation of nests of ground-nesting birds with precocial chicks is an issue, one approach in mitigation is the ‘headstarting’ of chicks, in which eggs are removed from nests soon after hatching and chicks reared in captivity and then released into the wild when the risk of nest predation has passed. Risks include lower subsequent survival than wild-raised fledglings and lack of socialisation from contact with adults, but initial results for wader species including curlew and black-tailed godwit appear positive (Colwell et al. 2020). Population effects of this practice are as yet unknown, but it involves intensive nest monitoring and animal husbandry work, so is likely to be costly.

Diversionary feeding, where food is used to draw animals away from problem activities or locations, has been used in a range of context in an attempt to reduce human-wildlife conflict, but has often not been evaluated for costs and benefits, with only five of 13 studies that allowed evaluation reporting success up to 2016 (Kubaciewicz et al. 2016). In the context of the management of predation impacts, it has been used and assessed in the context of hen harrier feeding on red grouse in Scotland, a system in which overall biodiversity is low, with fewer other predators and a less diverse prey base than is found in lowland farmland, for example. This has shown a reduction in red grouse chick predation, but there is no evidence yet about overall grouse productivity or population levels (although these will also be heavily affected by shooting) (Ludwig et al. 2018). Diversionary feeding has also not been trialled in the absence of other habitat management for grouse, is expensive and is likely to be far more difficult to target towards specific predators in other systems.

Major negative impacts on some predator populations have been achieved, historically, via nest destruction, poisoning and trapping/shooting of adults, notably in the case of raptors such as hen harrier, although details of the actions undertaken and their effects are not clear due to the illegality of the practice (Green et al. 1999, Etheridge et al. 1997). The expansion and population increase of red kite into Scotland appears also to have been slowed by such activities (Smart et al. 2010). However, most lethal control of predators concerns generalists such as carrion crow, magpie and red fox, as is legal in the UK. These generalists have not obviously been affected at the population level by long-term, large-scale removal efforts, but data on control activities, as well as on fox populations, limit inference. Long-term increases in many corvid populations, at least, suggest that control has at most slowed population increases. Reductions in control effort have been proposed as a driver of these increases (Gregory &Marchant 1996), but numbers have stabilised in the last decade or so (Woodward et al. 2020).

3.1.13.2 Co-Benefits and Trade-offs

None

3.1.13.3 Magnitude

Unknown, context-specific and highly variable.

3.1.13.4 Timescale

Initial responses of demographic rates may be within a year, population responses may take 5-10 years to appear, effort would need to be ongoing indefinitely. It is unknown how predators will respond to long-term control measures; changes in behaviour are likely under strong selection pressure and the efficacy of control measures is therefore likely to change.

3.1.13.5 Spatial Issues

Predator-prey systems are rarely closed (except on remote islands), so changes in local predator densities are likely to lead to redistributions and infilling within and/or between species. The scale over which such changes will occur are unknown and likely to be variable between species and contexts. This has implications for the spatial extent of management effort that is likely to be required to achieve desired outcomes, even where evidence bases are clear.

3.1.13.6 Displacement

Predator-prey systems are likely to be complex, with multiple predators interacting in the same space, constraining one another through various forms of competition. Hence, removing one predator could release another, with unpredictable effects on the wider ecosystem. Redistributions following measures in one area could also displace effects to elsewhere.

3.1.13.7 Maintenance and Longevity

Since predator-prey systems are rarely closed (except on remote islands), effort will need to be ongoing and indefinite. Passive measures such as fencing would need minimal maintenance, but probably ongoing annual checks and periodic repair. All active management activity, such as gamekeeping, nest-cages or headstarting, would have high costs and require continuous investment indefinitely.

3.1.13.8 Climate Adaptation or Mitigation

N/A

3.1.13.9 Climate Factors / Constraints

Predation regimes may change with climate, as distributions of predators and their prey bases change.

3.1.13.10 Benefits and Trade-offs to Farmer/Land-manager

Predator management may have positive effects on game shooting enterprises (although note that most released gamebirds will not be affected by numbers of nest predators, as they are released and shot while fully grown, in the same season). It is also possible that the mortality of new-born lambs might be reduced. Physical structures such as fencing and nest cages, as well as access for lethal control measures, may interfere with farm operations.

3.1.13.11 Uptake

Costs of most measures would be high, especially if applied at large scales, limiting feasibility. The skills required for gamekeeping may be in short supply. Large sectors of the public will disapprove of many lethal control measures for moral reasons and there would be lobbying against them.

3.1.13.12 Other Notes

None

3.2 BUNDLE: ACTIONS FOR HABITATS WITH SPECIFIC HYDROLOGICAL CHARACTERISTICS

Wetland and bog are not important habitats agriculturally, but cover large areas of the uplands and lowland fen landscapes and are farmed in various ways. For biodiversity – notably a set of habitat-specialist priority species that are particularly sensitive to climate change – these habitats are critical and vulnerable, with important potential effects of management involving drainage regimes, grazing and afforestation. System changes to or from wetland systems will inevitably have biodiversity consequences, although these will mostly not involve species that are identified as priorities in a farmland context.

3.2.1 EHAZ-137 Use paludiculture

Paludiculture is the practice of farming on wet land, such as rewetted bogs, water meadows and fens. Land is not drained and plants such as Sphagnum moss, bulrush Typha latifolia and reed Phragmites australis are grown as crops, with diverse uses (often not food). Biodiversity benefits of this system could potentially be all species that use bog habitats and not drained farmland. Conversely, farmland specialists would not benefit, but it is likely that the assemblage present in the managed land would be more like that found in the natural habitat in the site than it would if land were drained.

3.2.1.1 Causality

It is inevitable that returning areas of former wetland to a wetland or semi-wetland state, some wetland species will benefit, although these may not necessarily be priority species. It is recognised that paludiculture ‘may contribute to nature conservation objectives but might also contradict these’ (Greifswald Mire Centre https://www.moorwissen.de/en/paludikultur/hintergrund/hintergrund.php). However, it is possible to adjust paludiculture farming techniques, even where they involve intensive monoculture production, in ways that offer the potential to provide direct biodiversity benefits without significantly constraining agricultural practices or profitability, by following certain general principles (Mulholland et al. 2020). The evidence for this derives from studies of wetland management practices, as opposed to actual paludiculture operation, which are currently very rare, diverse and unreplicated in the UK (Mulholland et al. 2020), so effects at the scale and in the context of real, likely, interventions are not understood. This could influence speed of colonisation and population viability, for example. Nevertheless, the evidence from wetland management indicates relevant specific options to target biodiversity such as Bittern, characteristic spider communities, Orthoptera and ground beetles (Wichtmann et al. 2016, Mulholland et al. 2020), although these are likely to vary with crop type (Mulholland et al. 2020).

Potential indirect benefits of paludiculture for biodiversity have also been identified, since the management of true wetland nature reserves would be facilitated by their being surrounded by wet habitats, as opposed to drained land or forestry (Mulholland et al. 2020). Again, this would involve wetland specialist species. Paludiculture could be viewed as an intermediate stage between drained use and nature conservation, so could contribute through nutrient removal and vegetation management to the establishment of site conditions necessary to each conservation objectives. Paludiculture sites could also act as buffers or ‘stepping stone’ agents of connectivity between natural wet habitats (Greifswald Mire Centre https://www.moorwissen.de/en/paludikultur/hintergrund/hintergrund.php).

3.2.1.2 Co-Benefits and Trade-offs

Within biodiversity, it is clear than any habitat change would have a negative effect on the species that use the habitat in its original state. Here, this would be those associated with drained farmland, some of which are conservation priority species, such as Corn Bunting, Tree Sparrow and Yellow Wagtail, which have remaining population centres in former fenland in eastern England, for example. It is probably unlikely that paludiculture uptake would be sufficiently high to have significant effects on these species, and positive effects on them via invertebrate food abundance are possible (e.g. Field et al. 2008), but this is a potential problem to be considered.

3.2.1.3 Magnitude

This is currently unknown and unpredictable given the lack of examples in a practical context and at a known extent of application. It is also likely to vary hugely between species or taxonomic groups, e.g. it may be negligible for species that require large extents of habitat but high for those that can persist in small patches, if the scale of uptake is low.

3.2.1.4 Timescale

Timescales will vary with taxon, according to demography and speed of colonisation, e.g. spiders associated with early successional wetland habitat are likely to be found in paludiculture systems within 0-5 years (Mulholland et al. 2020. Species with low dispersal and currently restricted and/or declining populations, such as Bittern, would probably take more than 10 years to establish populations, even in ideal habitats.

3.2.1.5 Spatial Issues

Benefits are highly likely to depend on location and extent, especially for low-dispersal-ability species and those that require large extents of habitat. However, establishing paludiculture with sensitive management that is close to existing habitat and/or source populations is likely to maximise speed and size of response. Moreover, new wet habitat adjacent to existing patches may be valuable as a complement to those habitats and supporting species to persist or to expand in a habitat mosaic that would otherwise not be suitable.

3.2.1.6 Displacement

See 3.2.1.2 Co-benefits and trade-offs

3.2.1.7 Maintenance and Longevity

Paludiculture per se would require maintenance of water levels, in particular, but would be an active agricultural system and so would persist by definition. Specific management interventions to enhance biodiversity within the system would require annual or periodic implementation, such as mowing practices (Mulholland et al. 2020), so probably financial support on the same timeframe.

3.2.1.8 Climate Adaptation or Mitigation

As paludiculture fundamentally involves the management of water levels, any change to evapotranspiration, natural water levels or flooding regimes would affect the details of the management required, and hence viability (Mulholland et al. 2020). However, climate models ought to allow the risks to be built into plans for future roll-out.

3.2.1.9 Climate Factors / Constraints

Paludiculture would only be viable on former wetland, but could contribute to flood risk mitigation if implemented at sufficient scale, and both power use and greenhouse gas emissions from paludiculture are lower than from arable farming of similar land (Mulholland et al. 2020).

3.2.1.10 Benefits and Trade-offs to Farmer/Land-manager

Land-use trade-offs will depend on the economics of crop production versus alternative or existing cropping, but the enhancements of paludiculture to benefit biodiversity are reported to be cost-neutral (Mulholland et al. 2020).

3.2.1.11 Uptake

Barriers to the uptake of paludiculture associated with water level management (coordination with Drainage Boards, for example), weed control (weeds will not be known arable weeds, so may need new control systems), equipment for mechanisation, etc., are discussed in detail by Mulholland et al. (2020).

3.2.1.12 Other Notes

Given that paludiculture is likely only to cover small areas for the foreseeable future, it will not be monitored effectively by existing biodiversity monitoring schemes, so some bespoke monitoring would be required to assess effectiveness for biodiversity in practice.

3.3 Bundle: Climate measures

Climate change has already affected biodiversity in various ways, such as northerly shifts in distribution ranges, and further such changes are likely to occur as the climate warms. Management to mitigate the effects on sensitive species and/or to facilitate their adaptation to the changed conditions has the potential to minimise these effects, although their potential efficacy must be limited and dependent upon the scale of the change in climate that actually occurs. Measures in this area are specifically those concerned with biodiversity adaptation under a changing climate, but may also be critical for maintaining priority species and semi-natural habitats.

3.3.1 ECCA-032 Provide feeding/ breeding/ shelter and rest areas to support the lifecycles of species threatened by climate change

This 'action' actually conflates a wide range of individual potential interventions, which may be targeted at individual species or habitats. Various habitat creation and restoration options within Countryside Stewardship, for example, could implement relevant management. What constitutes a 'threat to a species' is also definable in multiple ways: for example, common species could decline to a lower population size reflecting reduced carrying capacity, species could cease to persist in some areas (but potentially colonise others), UK populations could go extinct, or global populations could go extinct. Feeding resources may be required if climate change is compromising existing food plants, for example, and could be provided by cultivating resilient varieties of these plants, or soil wetness allowing access to soil invertebrates, in which case the water table could be manipulated. Breeding, shelter and resting habitats are most likely to depend upon habitat (vegetation) structure, which could be maintained in the context of existing plants being replaced by climate-resilient alternatives, or by encouraging the establishment of the affected species in new, more suitable sites.

The most significantly affected species in a UK context are likely to be those at the southern edges of their ranges and those that are found in the uplands, but these will also be the most difficult species for which to create new habitats.

A full review of this 'action' is not feasible because of the breadth and lack of specificity of the interventions involved.

3.3.1.1 Causality

Given that climate effects are critical for particular species and that the life cycle stage(s) in which the effects occur have been identified accurately, then this type of intervention should be beneficial. However, this requires ongoing evaluation in the face of climate change effects for verification. It is not clear that there has been any relevant management and monitoring to provide relevant evidence to support the various interventions that may be being referred to here. Further, effectiveness will depend upon the actual degree of climate change that occurs.

3.4 BUNDLE: HABITAT CREATION

Priority species are often habitat specialists and the maintenance of priority habitats in the face of environmental pressures, including climate change, will also clearly require the creation of specific habitats, either from land under intensive production or through restoration. Therefore, effective habitat creation could be a key mechanism to allow population recovery and/or persistence. This creation could deliver extensions of existing semi-natural habitat, or novel habitats that provide resources for target species in different ways, such as within intensive landscapes (i.e. effectively replacing the value provided historically by less intensive agricultural systems, without compromising production). New habitat could be created specifically and only to support biodiversity, but it is likely to have other benefits as well, such as reducing nutrient pollutants, providing carbon sequestration or assisting with flood prevention.

3.4.1 ECAR-032 Create agroforestry systems

Agroforestry is a philosophy involving integrating the management of woody vegetation with farming of crops and livestock. Precise definitions vary, but can include the combination of hedgerows with crop and pasture fields, or the use of shelterbelts in farmland, as well as more novel systems integrating agriculture and silviculture. This review considers hedgerows and shelterbelts separately (see **ECAR-047** and **ECCM-025C**), so this section considers only silvoarable and silvopastoral forms of farming, in which cropping or livestock farming is integrated with forestry.

3.4.1.1 Causality

Torralba et al. (2016) conducted a meta-analysis on the effects of agroforestry on ecosystem service provision and biodiversity. There was a positive effect of agroforestry overall (effect size = 0.454, $p < 0.01$) relative to conventional agriculture and forestry, but this masked considerable heterogeneity. For biodiversity, all mean effects were positive, but they were significant only for birds (not for plants, fungi or insects). Erosion control, biodiversity and soil fertility were enhanced by agroforestry, but there was no clear effect on provisioning services and biomass production was affected negatively. Comparisons between agroforestry types and reference land-uses showed that both silvopastoral and silvoarable systems increase ecosystem service provision and biodiversity, especially when compared with forestry land. The Conservation Evidence review found mixed evidence for agroforestry benefits for bats, but all studies involved tropical systems (reviewed by the Conservation Evidence project2). Further, most other studies have also been conducted in the Mediterranean region, so direct relevance to the English context is uncertain. Effects on 'biodiversity' also consisted of a conflation of whatever metrics for species richness or abundance were found in individual studies, so reflected a gross assessment and probably masked a range of negative associations with individual species or species groups, as well as positive ones. In general, silvopastoral and silvoarable systems will generally increase habitat heterogeneity when introduced into intensively managed farmland, which will tend to increase species richness, but the concomitant loss of open habitat and addition of woody structure adjacent to remaining open habitat will have negative effects on various priority species that are associated with such landscapes.

The literature indicates that, compared with conventional agriculture, alley cropping systems (ACS) have the potential to increase carbon sequestration, to improve soil fertility and generally to improve resource utilisation. ACS may also help to regulate water quality, to enhance biodiversity and to increase overall agricultural productivity, with particular value for marginal land (Tsonkova et al. 2012).

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[2] https://www.conservationevidence.com/actions/963

3.4.1.2 Co-Benefits and Trade-offs

Silvoarable systems require fewer nitrogen inputs, both because the area of crop is reduced and because the greater litter input and more extensive root systems of the trees fix nitrogen in the soil, so they can improve soil nutrient management.

There may be a business resilience co-benefits from diversified income from trees, including high value tree and fruit crops in agroforestry systems.

Woody features will have hydrological benefits for flood mitigation when placed appropriately in catchments, i.e. along hillsides, but tree species differ in their effects on soil hydrology and water infiltration etc. (Webb 2021).

3.4.1.3 Magnitude

Effects on different taxa will be highly variable; scale of implementation will probably also have a large effect.

3.4.1.4 Timescale

Biodiversity and other benefits would begin to appear in years 0-5, for example from replacing ash trees, natural regeneration of hedgerow trees and development of agroforestry systems, and continue to develop over many years as the trees mature.

3.4.1.5 Spatial Issues

Negative effects of trees on open country biodiversity mean that insensitive siting of new agroforestry projects could have detrimental impacts on priority species such as skylark. Hydrological benefits depend on location within catchments. Benefits to the connectivity of woody habitats in the landscape will clearly depend on location with respect to existing woody features.

3.4.1.6 Displacement

There is likely to be some displacement of agricultural land, and associated crops and biodiversity, if large-scale agroforestry is introduced on improved land and the new system does not deliver equivalent yields.

3.4.1.7 Maintenance and Longevity

Agroforestry interventions can be permanent, if farmers perceive the benefits of trees, hedges and shelter belts and continue to manage these features for the long-term, replacing trees lost to disease, livestock damage or harvested. However, market support may be needed to make the land-use sustainable in the medium term.

3.4.1.8 Climate Adaptation or Mitigation

There is emerging evidence on the potential climate mitigation and adaptation benefits of agroforestry systems. It certainly has significant role in the decarbonisation of the UK economy, with sequestration benefits dependent upon the type of system and the soil. Kay et al (2019) show that strategic and spatially targeted establishment of agroforestry systems could provide an effective means of meeting objectives on GHG emissions whilst providing a range of other important benefits.

3.4.1.9 Climate Factors / Constraints

Tree crops are a long-term investment and must be sensitive to future climate change making conditions unsuitable (or sub-optimal) for chosen species. Establishment therefore needs to use the best climate predictions available and would ideally include contingencies for different climate scenarios occurring.

3.4.1.10 Benefits and Trade-offs to Farmer/Land-manager

Agroforestry systems may be more sustainable in the long term, given climate change and problems with the yields of established crops or dependence on chemical inputs. However, high establishment costs and time lag to crop delivery, especially from trees, will cause economic problems and the need to learn new land management skills could prove costly, hence requiring significant investment.

3.4.1.11 Uptake

There is likely to be considerable farmer reticence due to unfamiliarity with a new system, high establishment costs and time lag to crop delivery, especially from trees. Some systems require societal change to operate sustainably at large scales in the economy, such as wide acceptance of nuts as a protein source, reduced meat consumption and acceptance of an agroforestry premium for meat products.

3.4.1.12 Other Notes

None.

3.4.2 EHAZ-010X Create permanent grasslands

The creation of grassland from arable land has the potential to support species that benefit from undisturbed, uncultivated soil, as well as those that directly depend on grass. Further, some species with large home ranges benefit from the availability of arable and pastoral land adjacent to one another, i.e. mixed farming. All of these species may therefore respond positively to the addition of grassland to arable-dominated farming systems (‘arable reversion’). Options doing this have been implemented in various agri-environment schemes for decades now.

3.4.2.1 Causality

Arable reversion options have been reported to be very effective in establishing a sward and fitting with the farming system (Marshall et al. 2020). There is good evidence that adding grassland to arable-dominated landscapes has positive effects on local bird diversity, probably reflecting resource availability benefits at the landscape scale that indicate benefits for various other taxa, such as soil invertebrates (Siriwardena & Robinson 2002). In respect of a specific agri-environment intervention, arable reversion has been employed in English schemes from more than 20 years and grassland created by sowing traditional chalk grass species supported higher Skylark densities than permanent grassland created with agricultural grass species, although neither supported densities as high as undersown spring barley (Wakeham-Dawson et al. 1998). Further, grazing controls to allow a longer sward provide further benefits in reverted chalk downland (Wakeham-Dawson et al. 1998). Similar results were found considering winter bird abundance in the same study region in southern England (Wakeham-Dawson & Aebischer 1988). Raising water levels can also be used to revert drained arable land to wet grassland, i.e. grazing marsh, leading to plant communities that are typical of the target habitat and breeding wading bird population responses locally (Lyons & Ausden 2005).

Pywell et al. (2002) examined the creation of diverse grassland communities on ex-arable land in a multi-site experiment over a wide variety of soil types and locations throughout lowland Britain. They found that a species-rich seed mixture sown following deep cultivation produced vegetation communities most like specified target communities from the UK National Vegetation Classification. Natural regeneration and treatments sown with a species-poor seed mixture were much less similar to the target. The sites on circum-neutral soils achieved the greatest degree of similarity to the target. Those on calcareous and acid soils failed to achieve their targets and most closely resembled the target for neutral soils. This reflected the poor performance of the sown preferential species for these communities.

Woodcock et al. (2012) investigated changes in restoration success for butterflies for arable reversion sites, where grassland was established on bare ground using seed mixtures, and grassland enhancement sites, where degraded grasslands are restored by scrub removal followed by the re-instigation of cutting/grazing.

Consistent increases in restoration success over time were seen for arable reversion sites, with the most rapid rates of increase in restoration success seen over the first 10 years. For grassland enhancement, there were no consistent increases in restoration success over time. The results supported arable reversion is an effective tool for butterfly conservation, but with a time lag of several years, which needs to be accounted for.

Woodcook et al. (2010) found that increasing the similarity of the plant community at restoration sites, using spreading of green hay to introduce target plants, to target species-rich grasslands promoted restoration success for phytophagous beetles.

3.4.2.2 Co-Benefits and Trade-offs

Arable reversion to grassland might be expected to deliver increases in soil carbon, but performance in carbon sequestration may actually be poor, due to reduced available soil nitrogen (Gosling et al. 2017).

3.4.2.3 Magnitude

Effects may be significant for many groups and individual species, but will depend on scale, vary over time and are far too complex to be summarised here.

3.4.2.4 Timescale

Restoration success will involve a timelag of a decade or more for vegetation succession to complete.

3.4.2.5 Spatial Issues

Landscape context is important, with restoration being approximately twice as successful in those landscapes containing high as opposed to low proportions of species-rich grassland (Woodcock et al. 2010). By targeting grassland restoration within landscapes containing high proportions of species-rich grassland, dispersal limitation problems associated with restoration for invertebrate assemblages are more likely to be overcome.

3.4.2.6 Displacement

There is clear potential for arable production to be displaced, but this may be minimised by converting only low-productivity arable land. Pairing management with increased arable production elsewhere could also avoid a net displacement.

3.4.2.7 Maintenance and Longevity

A retrospective review of arable reversion retention found that a narrow majority (56%) of arable reversion parcels were retained following the end of agreement. This included the retention of whole-parcel options (37%) and retention of part-parcel options (19%). Long-term, continued support for management of reverted land, together with guidance for entering reverted land in to future agri-environment schemes may be critical for enhancing the maintenance of reverted land (Marshall et al. 2020).

3.4.2.8 Climate Adaptation or Mitigation

Adding grassland to an arable landscape could be significant in aiding connectivity of grassland areas and hence aid climate adaptation for some species. There is limited evidence for a carbon benefit.

3.4.2.9 Climate Factors / Constraints

Sown grass mixtures need to be able to thrive under future climates, so the best predictions should be taken into account.

3.4.2.10 Benefits and Trade-offs to Farmer/Land-manager

Loss of arable land and the need for grazing or mowing are likely to have management costs, and the availability of equipment, grazing animals and expertise may present additional barriers and hidden costs. However, a diversified farm business may offer increased resilience.

3.4.2.11 Uptake

Loss of arable land and the need for grazing or mowing are likely to have management costs, and the availability of equipment, grazing animals and expertise may present additional barriers and hidden costs.

3.4.2.12 Other Notes

None

3.4.3 ECCM-025C Plant hedgerows and ECCM-080C Plant hedgerows around point-source polluters

Hedgerows are characteristic of many traditional lowland farming systems, consisting of native shrubs planted along field boundaries and 'laid' to produce a continuous barrier. They frequently have an accompanying drainage ditch and lines of mature trees or individual trees within them. Creation of new hedgerow commonly involves filling gaps or restoring/recreating hedges along existing field boundaries, but may also involve creating completely new field edges.

3.4.3.1 Causality

Hedgerows and field margin vegetation affect the richness and abundance of flora, invertebrates and birds (Boatman (ed) 1994, De Snoo, 1999, Hinsley & Bellamy, 2000), and there is good evidence that the structure and form of a hedgerow and its management, in terms of differences in width, height, fenced buffer strips, frequency of cutting, etc., has a big effect on biodiversity (Haddaway et al. 2018). However, there is actually little direct evidence as to the effects of adding hedgerows to the landscape. Some species are negatively affected by hedges, as they provide vertical structure in otherwise open farmland and the species are adapted to open plains or steppe habitat; Skylark and Lapwing are the most prominent examples.

Early flowering woody hedgerow species such as blackthorn may partially fill a gap for resources for pollinators in spring (Dicks et al. 2015), but this will partly depend on hedgerow management which is also targeted by AES (Staley et al. 2012).

Note that hedgerows may facilitate the presence of undesirable species: one study fund that they had a significantly higher Woodpigeon nest density by more than threefold than woods (Inglis et al. 1995). This is an example of how hedgerows are likely to benefit generalist and edge-dependent species by adding more habitat to almost any landscape and will probably increase richness-based metrics where baseline habitats are open fields, albeit at a cost to some species that require the latter.

3.4.3.2 Co-Benefits and Trade-offs

There is some evidence for (small) hedgerow co-benefits for soil carbon stocks (Ford et al., 2019) and also for net greenhouse gas emissions.

There is a possible co-benefit of (wide) hedgerows for improving biosecurity against some livestock diseases by reducing transmission between stock in adjacent fields, but wide hedgerows can also provide habitat for alleged secondary vectors (badgers).

There is evidence that hedges can be especially effective for interception of aerial pollutants, especially in urban/peri-urban environments and along roadsides (Morakinyo et al., 2016; Abhijith et al, 2017; Abhijith et al, 2019).

3.4.3.3 Magnitude

Effects could be large, especially at the field scale and where hedges are rare in the landscape. The strongest positive effects will be on species of semi-natural habitats, as they persist in farmland. True farmland specialists are likely to be affected less (and some negatively), but effects will be highly variable and cannot be generalised.

3.4.3.4 Timescale

Hedgerow creation or restoration is a long-term process, with habitats likely to take decades to mature to provide their full biodiversity value. Hence, existing evidence on the value of hedgerows from agri-environment interventions from the 1990s onwards deals with spatial comparisons of different habitat types or management activities. However, some benefits from the new woody vegetation may appear within a few years, as habitats are created for invertebrates.

3.4.3.5 Spatial Issues

Adding hedges to landscapes will have more effect, positive and negative, where there are fewer other hedges present already. There is a common misconception that hedgerows are effective corridors between fragments of woodland habitat, yet there is a lack of clear evidence of the positive benefits of hedgerows in increasing landscape connectivity for woodland-dependent taxa (Davies & Pulllin 2006, 2007) although there is good evidence of benefits of hedgerows for a different set of species (broadly, described as 'edge specialists'). However, at the landscape scale, there is good evidence for the importance of hedgerows to vertebrates, as navigational aids and for commuting between breeding and foraging sites. There is, however, comparatively little evidence that connected hedges are important corridors for animal dispersal. This is particularly true for invertebrates, although they probably do have a facilitating role to play in this respect (Wolton et al, 2013). Direct tests of the effects of hedgerow habitat creation on connectivity at a landscape scale have yet to be conducted. At the population level, connectivity effects are very difficult to prove because significant movements may need to occur only very occasionally. Nevertheless, influences on connectivity clearly depend on the locations of habitats that are present to be connected.

3.4.3.6 Displacement

There should be minimal effects as hedgerows are usually created on existing field boundaries. Enhanced (higher, wider) hedges may have negative effects the productivity of very adjacent land by shading, but such parts of fields are likely not to be highly productive anyway.

3.4.3.7 Maintenance and Longevity

Hedges can be permanent, if farmers perceive the benefits of trees and hedges, and continue to manage these features for the long-term, replacing trees lost to disease, weather or livestock damage. Hedge maintenance needs to be regular, annual or biannual (see **ECCM-025EM**).

3.4.3.8 Climate Adaptation or Mitigation

Hedgerows as agents of connectivity could play a role in climate adaptation, but the uncertainty about their true role in this regard (see above) should be taken into account. For example, they may provide effective functional connectivity (as stepping stones in the landscape), even when not delivering explicitly structural connectivity. However, while they might not be as effective as new habitat as larger tracts of woodland or shrubland, for example, they do provide a novel habitat resource and hence would the population size and resilience of species that are present: increased habitat quality and quantity support climate adaptation in this way (Hodgson et al. 2010).

3.4.3.9 Climate Factors / Constraints

None.

3.4.3.10 Benefits and Trade-offs to Farmer/Land-manager

Hedges, especially thick or tall ones, can have negative effects on farm operations, making vehicular access more difficult and shading crops. Ideal hedgerow management dates may fall at times of the year when it is not convenient to farmers, e.g. when fields are wet, so accessing boundary vegetation for cutting may mean damaging the ground. This may limit uptake where soils are heavy or rainfall is high.

3.4.3.11 Uptake

Real or perceived negative effects on farming operations and productivity can be reasons for hedgerows not to be introduced; the same factors that led to hedgerows being commonly removed and left unmanaged since the 1970s will still apply. Ideal hedgerow management dates may fall at times of the year when it is not convenient to farmers, e.g. when fields are wet, so accessing boundary vegetation for cutting may mean damaging the ground. This may limit uptake where soils are heavy or rainfall is high.

3.4.3.12 Other Notes

None.

3.4.4 ECPW-272 Convert field horticulture to greenhouse or aqua/aero/hydroponics horticulture

Field horticulture is not a major land-use in England, but does cover significant areas of farmland in areas such as Lincolnshire. In these arable landscapes, it may provide valuable habitat heterogeneity, especially as broadleaf, spring cropping in farmland that is dominated by winter cereals. This heterogeneity can support bird nesting, for example, through a longer season by providing greater access to bare ground, and the microclimate below broadleaf crop plants will support different invertebrates than that under cereals. However, horticultural crops may also be grown in short rotations, requiring frequent tillage, and be subject to intensive chemical management, which are both likely to limit the biodiversity benefits.

3.4.4.1 Causality

There has been no research into the biodiversity effects of horticulture conversion or monitoring of ongoing processes. Conversion of field horticulture to ‘indoor’ approaches could remove some valuable habitat, but the effects are likely to be limited in scope and significance for populations. If the conversion creates space for alternative land-uses, there could be positive effects, but these clearly depend on the nature of the replacement land-use.

3.4.4.2 Co-Benefits and Trade-offs

[TOCB Report-3-3 Soils] Field horticulture production systems can impact significantly on soil health and the extent of erosion. Growing horticultural crops hydroponically could therefore have moderate positive benefits for soils.

3.4.4.3 Magnitude

The magnitude of any effects is likely to be small, as the spatial footprint of horticulture in the UK is small and localised.

3.4.4.4 Timescale

Any effects of system change on biodiversity would be rapid (within a year), as the habitat change to another land-use is likely to change the resources available to species in the habitat immediately, or as the crop or other vegetation matures.

3.4.4.5 Spatial Issues

This action is limited to current areas of horticulture by definition.

3.4.4.6 Displacement

The effects of this management action critically depend on the replacement land-use. Buildings on the same land or on other land with habitat value elsewhere could well lead to a net negative effect. However, a reduced spatial footprint, combined with replacement by habitats that provide more or better resources, could lead to net positive effects on particular taxa.

3.4.4.7 Maintenance and Longevity

The new form of horticulture would need to be economically viable to be sustained, or land-use would presumably change again, but otherwise this change in use would be a discrete event requiring no ongoing management.

3.4.4.8 Climate Adaptation or Mitigation

Climate should not affect this action directly, but could affect the economic viability of different forms of horticulture.

3.4.4.9 Climate Factors / Constraints

Climate should not affect this action directly, but could affect the economic viability of different forms of horticulture.

3.4.4.10 Benefits and Trade-offs to Farmer/Land-manager

The economics of different forms of horticulture – infrastructure, inputs, markets, etc. – will determine profitability and the attractiveness of the system change in the context of different payment rates.

3.4.4.11 Uptake

The economics of different forms of horticulture – infrastructure, inputs, markets, etc. – will determine profitability and the attractiveness of the system change in the context of different payment rates.

3.4.4.12 Other Notes

None

3.4.5 ECCA-018C Plant large-scale woodland in priority catchments

A range of priority species in the UK depend on broadleaf woodland habitat and species with larger home ranges or lower dispersal ability benefit more from larger-scale habitat creation. The location of this woodland in respect of catchments is not likely to be critical, so the key factors are the fact that woodland is being created and the scale of the new habitat patches. It is assumed that this action relates to broadleaf woodland only and that planting would consist of native trees.

3.4.5.1 Causality

Woodland has distinct, specialist flora and fauna, as well as supporting more generalist species. Woodland creation involves a complete change in habitat coverage, so impacts on species presence and abundance are inevitable. The spatial coverage of woodland (the natural climax vegetation in the majority of the landscape) in England was subject to large historical declines up until some increases with planting regimes in recent decades, leaving a generally fragmented habitat at the landscape scale, with many more-or-less isolated woodlands that are separated by open farmland or upland. Therefore, as a general rule, there will be positive effects on a range of woodland species from the planting of new woodland to deliver both larger habitat areas and enhanced connectivity of the existing patches. The challenge with collecting direct evidence of these effects is that woodland takes many decades to mature and the most specialist species tend to require large tracts of mature forest. Effects are also likely to vary with constituent tree species. Nevertheless, there is now good evidence for positive effects on some priority bird species of new woodland planting in a farmland context over a 30-year timescale, although with significant effects of location relative to other woodland habitat in the landscape and locally (Dadam et al. 2020). Research considering created mature woodland has yet to be conducted because of the inevitable decades of time lag. However, farm woods of around ten years old attract scrub, hedgerow and opencountry bird species (Vanhinsbergh et al. 2002), while the same woods, at c.30 years old, attract more woodland species, but with rather little difference in total species composition (Dadam et al. 2020). Differences in community structure from the 1999 survey were small, but Simpson's diversity was marginally higher and the more mature habitat in 2019 supported higher densities of 37 species, but lower ones of 23 species. Many decreasing species were those more associated with scrub or open habitat, but also included species that have declined nationally (Dadam et al. 2020). In a specific Welsh AES context, woodland establishment under Tir Gofal was tested for effects on bird population growth rates by Dadam & Siriwardena (2019), showing mixed effects. Of ten mostly generalist species, there were significant or nearsignificant effects for five species, of which three were positive. Note that these woodlands would have been less than 20 years old at the end of the period of evaluation.

The creation of new woodlands on farms is likely to increase the local population of a number of pest species (e.g. rabbit, *Oryctolagus cuniculus*; grey squirrel, *Sciurus carolinensis*) including the woodpigeon (*Columba palumbus*) (Inglis et al. 1995).

Moore et al. (2003) found that small mammal numbers were far higher in newly planted woodlands than in both hedgerows and agricultural land. Seven species were trapped in farm woods compared with five in hedgerows and two in agricultural land and the most frequently trapped species in the establishing woodlands was the harvest mouse *Micromys minutus*, followed by the wood mouse *Apodemus sylvaticus*.

Woodland plant communities (understorey and canopy) appear able to assemble spontaneously in 40 years in some circumstances (e.g. Harmer et al. 2001), but specialist, poorly dispersing and rare species are likely to remain absent for extremely long periods especially where legacy effects of disturbance and increased soil P and N levels persist (Dupouey et al. 2002; Strengbom et al. 2001; Naaf & Kolk 2015). Woodland flora has been found to be richest before canopy closure, with steady development of ground flora but substantial turnover (Harmer et al. 2001). Assembly of 'typical' or 'desirable' woodland understoreys will then often require overcoming poor inherent dispersal and long distances to source populations plus land-use legacy effects. The latter include soil seedbanks, in situ vegetation and high soil fertility, all strongly influenced by previous agriculture (Coote et al. 2012; Harmer et al. 2001).

Research on habitat creation and management for pollinators has tended to focus on open habitats, e.g. meadows and arable field margins under agri-environment schemes (AES), rather than woodland. Woodland creation may impact pollinators during early stages, but a climax or equilibrium state may not be achieved even after decades. As such, the overall effects of woodland creation on pollinators are difficult to assess within the duration of the average research grant. This may partly explain the lack of clear evidence on the subject. There is a need for more experimental studies observing the impacts of woodland creation and management on pollinators *over* longer time periods. For cost-efficiency, such experiments may be best established at the same time as woodland creation initiatives.

There is limited evidence on the nuances of where woodland creation or management are best placed to benefit pollinators. However, internationally, studies have quantified the scales over which woodland can increase pollinator abundance in surrounding agricultural landscapes.

It would be anticipated that woodland expansion will directly affect many aspects of habitat quality for woodland mammals. Of the few studies which have investigated such broad scale land use changes, only zero to small positive benefits for woodland mammals have been indicated as a result of woodland expansion. Despite differences in mammal diversity not being detected between plantations and other habitats (Stephens & Wagner 2006), greater mammal abundance (but not species richness) was detected by Felton et al. (2010) in plantations when compared to land purely composed of pasture, and by Moore et al. (2003), for small mammals in newly planted woodlands on farmland compared to both hedgerows and agricultural land. However, no clear positive effects on woodland mammals were detected when woodland expansion was followed over eight years (Lindenmayer et al. 2008).

Negative effects of woodland creation can involve a wide range of outcomes, including agricultural productivity, landscape cultural value, elements of biodiversity and soil carbon (on some soil types). Woodland creation can pose risks to pollinators, especially to species associated with open semi-natural habitats. Negative effects of woodland creation could result if woodland form obstacles to bees foraging in open habitats, for example (Goulson et al. 2010). As such, managing risks to pollinators involves understanding the starting point of woodland creation; on species-rich wildflower meadows, woodland creation is likely to have little benefit or even detrimental results for pollinators.

Woodland creation could increase the abundance of diseases, pests, predators and competitors, some of which are non-native. Outcomes for pollinators are difficult to predict due to the complexity of possible species interactions. Risks could be exacerbated if goods and/or trees are imported for woodland creation and management. One possible risk would be increases in the Asian hornet *Vespa velutina*. This species has been repeatedly sighted in Southern England in recent years and poses a threat to honeybees. It is possible that honeybees themselves act as non-native competitors to native pollinators. However, one large international study found that pollination by managed honey bees supplemented, rather than substituted for, pollination by wild insects (Garibaldi et al. 2013).

Potential negative effects of woodland creation on birds relate to non-woodland issues: (a) the habitats that are replaced and (b) effects of the new habitat boundary or heterogeneity that is created. With woodland creation on farmland or moorland, the replaced habitat typically has low biodiversity value (few species present) and many species associated with boundary habitats are also found in woodland. Increased heterogeneity at the landscape scale is also often a positive influence on species abundance, although not for all species, and for community diversity as a result (Pickett & Siriwardena 2011). However, woodland creation facilitates the presence of predators in otherwise treeless landscapes, with potential negative effects on other species. This can be due to real or perceived predation risk, as per the common avoidance of tall habitat structure by ground-nesting birds (e.g. Chamberlain & Gregory 1999). The principal evidence for negative effects on population viability involves upland-nesting wading birds and plantations (Amar et al. 2011, Douglas et al. 2014). This indicates that careful consideration of geographical context is important to minimise impacts on sensitive species of conservation concern.

3.4.5.2 Co-Benefits and Trade-offs

Woodland planting in most contexts will deliver net carbon sequestration, at least until felling. Woodland can also decrease flood risk, especially with strategic planting in catchments (Carroll et al. 2004). Woodland can provide shelter, hence protection from wind erosion, and trees can enhance air quality. However, proximity to open habitat would lead to negative effects on several species that use such habitats.

3.4.5.3 Magnitude

The effects of this action will be large – woodland, particularly high-quality woodland, is not a common habitat in many English landscapes – but also variable because species vary in their dependence on woodland, as opposed to other patches of semi-natural habitat. Negative effects are likely to be highly localised.

3.4.5.4 Timescale

Time lags, especially for specialists, are very long, so a focus of multiple decades is required, opening a potential issue with unknown interactions with climate change. However, there is evidence for medium-term benefits for birds. Likely timescales for dispersal of typical or desirable woodland plants into newly created woodland vary with climatic region, the favourability of the intervening matrix separating source and recipient woodlands (Svenning & Skov 2002) as well as distance to source populations. Long-distance dispersal and establishment events can also occur but are often rare.

3.4.5.5 Spatial Issues

Location could be critical for woodland creation effects, due to the function of the habitat as stepping stones between existing woodland patches, i.e. the broader landscape context. There are also negatives for certain species (e.g. breeding waders) if woodland is close to sensitive open habitat due to the facilitation of predation.

Kimberley et al. (2014) found that spatial factors acted to filter immigrant plant species non-randomly. Isolated plantations were more likely to be colonised by well-dispersed species (adhesive or wind-dispersed seeds), while only plantations contiguous with existing ancient woodland increased in plant species richness to the levels of the adjacent woodland. This took 70-80 years to occur. Jacquemyn et al. (2003) also showed that vascular plant species richness was significantly lower in recent woodlands greater than 100 m from long-continuity woodland compared to recent woods adjacent to long-continuity woodland. Humphrey et al. (2015) summarised relevant literature comprising 28 spatial or temporal studies of vascular plant diversity responses to abiotic, temporal and spatial factors. They showed that while patch characteristics, that is abiotic and biotic conditions within the woodland, were important in 88% of studies, an effect of surrounding habitat was important in 80% and isolation in 74%.

Results from Kimberley et al. (2015) and Kolk & Naff (2015) suggest that newly planted woodland or increases in extent of existing woodland encouraged by natural expansion in the absence of grazing, should focus on existing long-continuity woodlands or with a larger persistent species pool of woodland plants but where these woodlands have seen reduction in historical extent. Harmer et al. (2001) suggested that colonisation and plant community assembly is also likely to be more rapid if existing linear features are included, for example hedges rich in remnant forest species.

New woodland location and characteristics have significant effect on colonisation and use by birds. Dadam et al. (2020) characterised woodland connectivity in terms of areas (weight of surrounding habitat) and numbers of nearby patches (numbers of point sources), but these were fairly highly correlated, so the potential to discriminate between these effects was limited. The connectivity analyses showed that, although patterns varied between species, woodland connectivity generally had a negative effect on abundance at the local scale, but a more mixed effect (and often a positive one for specialists) at the landscape scale. This suggests that the use of farm woodland patches by birds during their daily activity is lower where there is more nearby woodland, possibly because this habitat is more mature and provides better or more resources. Conversely, where there is little surrounding woodland locally as an alternative source of resources, perhaps birds use farm woodlands more. At the landscape scale, there is then some evidence that specialists (in particular) are more likely to colonise new woodland plots in more heavily wooded landscapes. Adding woodland to less wooded areas at the local scale that have more broadleaf woodland and less coniferous woodland at the landscape scale is likely to deliver larger local populations.

More complex plot shapes (longer perimeters per unit area) were associated with lower abundances for 33 species, including 12 specialists (Dadam et al. 2020), suggesting that there was no strong preference for edge habitats across the assemblage, although deviation between edge and core habitats would be expected to be greater in mature woodland.

Whytock et al. (2017) considered birds in a wide range of woodland ages and found that local patch characteristics were relatively more important than landscape characteristics for bird communities, and biodiversity responses to habitat creation depended on local- and landscape-scale factors that interacted across time and space. Dolman et al. (2007) reviewed the evidence for patch area and composition effects on woodland birds globally, finding that larger woodlands support more woodland bird species, and that woods located within sparsely wooded landscapes are less valuable to specialist woodland species. Species found in small woods generally also occur in large woods, but small woods may be preferred by a few edge species and are more variable in bird assemblage composition. They concluded that the metapopulation dynamics of specialist species with poor dispersal (typically those of most conservation concern) shows that creating or buffering large woodlands is more efficient than a greater total area of small fragments. Connectivity among smaller fragments appears to benefit widespread generalist species. However, this study considered patterns among contemporaneous, mature woodland, not among newly created habitat, so the conclusions would relate to a hypothetical, very long-term context with no gross environmental change, if they were applied to inform new woodland planting. This evidence should therefore be weighed against that from direct studies of woodland creation, where there are apparent conflicts in consequent recommendations for best practice, considering the timeframe and species range that are of interest for target-setting.

There is clear evidence that effects of woodland on crop pollination are distance-dependent. For example, Joshi et al. (2016) observed that positive effects of proximity to woodland on flower visitation were apparent within 500m. Similarly, Bergman et al. (2018) found that, for the majority of the 30 most common butterfly species in their study, there were strong positive responses to the amount of forest cover within 200–500m (although an earlier study only found effects at scales >2km; Bergman et al. 2004). Diaz-Forero et al. (2011) found that some bumblebee species responded positively to forest at small spatial scales (250-500m) while other species responded negatively at large spatial scales (1-2km). Ricketts et al. (2008) found that visitation rate declined more steeply than pollinator richness with increasing distance from natural or semi-natural habitats (half maximum at 0.6km and 1.5km respectively). Garibaldi et al. (2011) observed continuous declines in visitation and fruit set up to >3km from natural habitats. In general, when creating or restoring any habitat, the existence of source populations will limit colonisation potential (Hanski 1994), particularly for rare and less mobile species.

3.4.5.6 Displacement

Any loss of open habitat, notably farmland, is likely to lead to farming activity potentially being moved elsewhere, although this would be minimised by siting in areas of low agricultural productivity. Note that such an approach at the national scale would produce a very skewed distribution of new woodland, with little being added in regions such as the south and east, or lowlands.

3.4.5.7 Maintenance and Longevity

There is some anecdotal evidence of new farm woods with a 30-year maintenance stipulation being removed after that 30-year period, so indefinite financial support for landowners might be helpful, if practicable.

Disease issues such as ash dieback may mean that active replanting is necessary in the future and, in general, woodland benefits are likely to be maximised by ongoing active management, so transitioning from woodland *creation* support to woodland *management* support. Similarly, long-term measures such as fencing to control deer browsing may be necessary.

3.4.5.8 Climate Adaptation or Mitigation

Woodland creation has clear potential contributions to climate change mitigation and adaptation by providing new habitat and connecting existing habitat, as well as constituting a carbon sink.

3.4.5.9 Climate Factors / Constraints

Trees need to persist under local climates decades into the future, so the choice of species and varieties needs to consider the best predictions of future conditions.

3.4.5.10 Benefits and Trade-offs to Farmer/Land-manager

Loss of farmed land, shading of crops, and facilitation of problem species (e.g. rooks) could all be negative influences for farming. Conversely, there could be new business opportunities with woodfuel and other woodland products.

3.4.5.11 Uptake

Payments may need to be very long term – some Farm Woodland Scheme woods were chopped down after the 30-year scheme requirement ceased. Loss of farmed land, shading of crops, and facilitation of problem species (e.g. rooks) could all be negative influences for uptake by farmers.

Staddon et al. (2021) conducted a thorough review of the social and behavioural factors involved in farmers' decisions regarding the expansion of tree cover on farmland. They found that decisions are influenced by farmer attitudes, values, skills, business aims, market drivers and social pressures, with farmers more likely to plant trees where scheme objectives align with their own objectives. They must also be able to perceive local on-farm benefits (e.g. soil protection, livestock welfare, biodiversity) and/or receive financial reward for tree planting on their farm. 'Indirect' and long-term, global benefits involving climate change mitigation are not sufficient alone. In order to achieve large-scale woodland expansion, behavioural changes reflecting changes in farmers' and social norms are needed. This requires starting with those most likely to engage tree-planting and aiming to shift social norms such that it is accepted practice amongst a wider set of land managers in due course.

3.4.5.12 Other Notes

None

3.4.6 ECPW-071 Create, enhance or manage floodplain woodland and ECPW-071C Create floodplain woodland

Creation of floodplain woodland is likely to have similar effects on biodiversity to creation of other woodland (see action **ECCM-048**), although benefits of large-scale woodland creation, i.e. support for species with large home ranges or low dispersal (meaning that viable populations need to be able to persist within a single habitat patch), may not be delivered. Enhancement and management of woodland is considered under actions **ECPW-071EM** and **ECCA-018EM**. Only aspects of the floodplain woodland context that are not considered in other sections are, therefore, discussed here.

3.4.6.1 Causality

See other woodland sections.

3.4.6.2 Co-Benefits and Trade-offs

Floodplains may feature wet grassland habitats with conservation value, notably including breeding wading birds. Woodland in floodplains may reduce the quality of adjacent grassland for these species, either because they choose to avoid habitats close to such vertical structure, or because they are directly negatively affected by predation from predators that use the woodland for shelter, cover or vantage points.

3.4.6.3 Magnitude

This is entirely dependent on the scale of woodland planting, but probably limited because woodland on floodplains is unlikely to be large in scale.

3.4.6.4 Timescale

See other woodland sections.

3.4.6.5 Spatial Issues

See other woodland sections.

3.4.6.6 Displacement

Existing floodplain habitat is likely to be grassland or arable, but either may be high-quality agricultural land, although if wet grassland, it is likely to have limited agricultural value, but potentially high biodiversity value. Therefore, establishing woodland is likely to displace land-use that is of value for agriculture or biodiversity.

3.4.6.7 Maintenance and Longevity

See other woodland sections.

3.4.6.8 Climate Adaptation or Mitigation

See other woodland sections.

3.4.6.9 Climate Factors / Constraints

See other woodland sections.

3.4.6.10 Benefits and Trade-offs to Farmer/Land-manager

See other woodland sections.

3.4.6.11 Uptake

See other woodland sections.

3.4.6.12 Other Notes

None.

3.4.7 ECPW-156 Plant, enhance or manage trees and shrubs

This action is interpreted as considering the planting of trees and shrubs outside woodland, i.e. as individual trees or shrubs, or small patches thereof. It is thus equivalent to **ECCM-024EM** Plant or manage trees outside of woodlands, including shelterbelts. This management action could have benefits for multiple facets of biodiversity, chiefly through increasing habitat heterogeneity and introducing semi-natural vegetation into production-dominated landscapes. It is likely to be used in field boundaries and unproductive field corners, so to have minimal effect on farm production.

3.4.7.1 Causality

Trees and shrubs outside woodland provide nest sites for birds, stepping stones for connectivity between woodland patches and habitat for species that require woody substrates or specific tree species. Negative effects could be the support for generalist predators, such as Carrion Crows, that would otherwise be unable to nest. The habitat is also likely to support species that use hedgerows and a combination of field boundary and open field habitats, which are more likely to be common generalists than habitat-specialist priority species, which require larger habitat patches, although do include some farmland specialists of conservation concern.

Small woodland patches have limited ecological value compared to large or well-connected woodlands (Dadam et al. 2020), and will allow increased Woodpigeon densities (Inglis et al. 1995). There is some literature on trees outside woodland, but this naturally considers mature trees, so is not necessarily directly relevant to tree/shrub planting and its effects in the short-to-medium term. Brown & Fisher (2009) produced a comprehensive review of trees outside woodlands (TOWs). They found that the habitat value of TOWs depends critically on their species, landscape context and group size. They are likely to be a valuable resource for many species in those areas with low woodland cover, such as regions of intensive agriculture and urbanisation. In well-wooded landscapes they act as corridors and stepping stones that increase the permeability of the landscape and contribute to the total area of edge or transitional woodland habitat. They found that there is ample evidence in the research literature that isolated trees and copses enhance biodiversity above those levels found in the surrounding matrix, either because they are relics of a former ecosystem or because they provide resources not available elsewhere in the landscape and so attract species. Brown & Fisher (2009) found many studies globally that highlight TOW importance for biodiversity, but few in the UK, although covering a wide range of groups. Whilst the communities associated with TOWs may be qualitatively different from those found in continuous woodland, they often contain a significant component of woodland specialist species.

3.4.7.2 Co-Benefits and Trade-offs

Limited carbon benefit.
Shading, facilitation of predation of priority species.
Landscape character is changed by large-scale woodland planting.

3.4.7.3 Magnitude

Patch sizes are small by definition, so effects are likely to be small, but will be larger (in terms of both positive and negative effects) in landscapes where trees are rare.

3.4.7.4 Timescale

Effects of planting trees and shrubs would take decades to be realised, but those of management and enhancement are likely to be realised within a few years.

3.4.7.5 Spatial Issues

Habitats whose value depends on an open aspect, e.g. grassland habitats for ground-nesting birds, would be negatively affected by the introduction of trees in adjacent locations, due to the facilitation of predation. Therefore, this action might avoid such landscapes and perhaps focus on those where additional trees could have positive effects on connectivity for species that are associated with woody features.

3.4.7.6 Displacement

This will be minimal because small patches have a low spatial footprint and planting will usually, in practice, be conducted on poor land.

3.4.7.7 Maintenance and Longevity

Small patches and new trees may need fencing to protect them from deer browsing, with the fencing itself then needing maintenance. Otherwise, maintenance requirements are low but trees may also need protection from spray drift and agricultural operations, such as by buffering.

3.4.7.8 Climate Adaptation or Mitigation

There may be benefits to climate adaptation via landscape connectivity.

3.4.7.9 Climate Factors / Constraints

Tree species chosen for planting need to be resilient to long-term climate change.

3.4.7.10 Benefits and Trade-offs to Farmer/Land-manager

Trade-offs will be minimal because of the low spatial footprint and placement on poor land.

3.4.7.11 Uptake

Nothing significant.

3.4.7.12 Other Notes

None

3.4.8 ECAR-033C Create shelter belts (tree, woodland, scrub, and hedgerow) with appropriate species composition near sensitive habitats, ECAR-047 Create/enhance/manage shelter belts (tree, woodland, scrub, and hedgerow) with appropriate species composition on hill slopes and ECPW-080C Create wind breaks

Shelter belts and wind-breaks are interpreted as linear woodlands up to tens of metres across. Their primary purpose is for shelter of agricultural land from wind, rain and erosion, but, as woodland in the agricultural landscape, they also represent potentially important habitat heterogeneity and an intrinsic habitat resource for biodiversity. Management and enhancement of shelter belts will have similar effect to the same types of action in other woodlands (see actions **ECPW-071EM** and **ECCA-018EM**), but their shape and location may have specific gross effects, which are reviewed here.

3.4.8.1 Causality

Shelter belts support biodiversity in the same way as other woodland, but to an extent that will be limited by the narrow, linear nature of the woodland plots. Therefore, specialist, priority woodland species will probably not be supported significantly. However, they can and do support a range of species from multiple taxa, although (also as with woodland in general) they need to be subject to ongoing management to maintain their value to biodiversity, as well as the potential provision of ecosystem services to the surrounding farmland (Dix et al. 1995, Kujawa 2002, Lavrov et al. 2021). Shelterbelts may also support many of the same species that also use hedgerows, i.e. often those that also make use of in-field, adjacent habitats, although the suitability of the two habitat types for individual species will vary (Hinsley & Bellamy 2000). The loss of shelter belts from open landscapes such as moorland can remove nesting opportunities for species like Carrion Crow *Corvus corone* (often regarded as undesirable due to possible impacts on other breeding species), but also those for priority species such as Merlin *Falco columbarius* (Barker et al. 2017) and goshawk (Henderson & Conway 2017. Bats commonly forage along woody linear features and shelter belts were positively selected by two of four species in one UK study (McHugh et al. 2018).

3.4.8.2 Co-Benefits and Trade-offs

These will be similar to those of woodland creation, although are likely to differ in magnitude:
Reduction of flood risk (Carroll et al. 2004);
Wind protection from erosion;
Air quality;
Carbon;
Proximity to open habitat would lead to negative effects on several species that use such habitats.

3.4.8.3 Magnitude

Effect sizes will be limited as woods are not large, but negative effects on open habitat species could be disproportionate.

3.4.8.4 Timescale

Effects of planting trees and shrubs would take decades to be realised, but those of management and enhancement are likely to be realised within a few years.

3.4.8.5 Spatial Issues

Proximity to open habitat would lead to negative effects on several species that use such habitats. The orientation of shelterbelts in the landscape is clearly critical to their function providing shelter, but could also constrain ideal placement for other benefits.

3.4.8.6 Displacement

This is probably of low impact: shelterbelts will cover some agricultural land, but probably would only be used where risks to of condition of the land make planting economically beneficial.

3.4.8.7 Maintenance and Longevity

As with other woodland, shelterbelts will need fencing and/or ongoing active woodland management to maintain their value.

3.4.8.8 Climate Adaptation or Mitigation

Shelterbelts may assist with adaptation and mitigation measures thanks to habitat creation

3.4.8.9 Climate Factors / Constraints

Tree species need to be proof to likely future climate in the focal location.

3.4.8.10 Benefits and Trade-offs to Farmer/Land-manager

Shelterbelts will presumably protect land, soil, livestock, etc. from deleterious weather effects, countering the potential negative of lost production land.

3.4.8.11 Uptake

N/A

3.4.8.12 Other Notes

None

3.4.9 ECCM-024EM Plant or manage trees outside of woodlands, including shelterbelts

This action is interpreted as referring to trees outside woodlands, with shelterbelts included in the definition of woodland. Therefore, it is equivalent to **ECPW-156** Plant, enhance or manage trees and shrubs (section 3.4.7), which is discussed in full.

3.5 BUNDLE: MAINTENANCE AND RESTORATION OF CULTURAL HERITAGE SITES

3.5.1 EBHE-080 Remove Scheduled Monuments or heritage assets on the shine database that are not Listed Buildings or Scheduled Monuments from cultivation

Create/ manage buffer strips around heritage assets on the shine database that are not Listed Buildings or Scheduled Monuments or around Scheduled Monuments (link boundary features) on cultivated land Ecologically, this is equivalent to the actions considered in 3.1.2, so the action is not reviewed further here.

3.6 BUNDLE: MONITOR, PLANS, DATABASES, CONSULTATION AND RESULTING ACTION

Monitoring is an essential part of successful management action as well as being important for verification of effects and attribution of change to an action. Monitoring ahead of action can support targeting of effort where it is most needed or likely to be most effective, and therefore increase efficiency. Monitoring following action should support the feeding back of evidence into action design: even with the best evidence bases, unforeseen results are possible and ongoing revisions to action design are critical to maintain long-term success of interventions. However, monitoring per se can never deliver impact; it needs to be followed by informed actions.

3.6.1 ECCM-058 Monitor health of trees

Tree disease is an important environmental problem, both for trees themselves and for the species and ecosystems that depend upon them. Dutch Elm Disease, Ash Dieback and Sudden Oak Death are historical and current examples of diseases with large-scale effects. Monitoring to find such effects early and hence to support action to mitigate of their effects would be highly valuable.

3.6.1.1 Causality

Monitoring is an essential element of successful control of tree disease but does not inevitably lead to successful control. The development of control measures is an active research area for diseases such as Ash Dieback and Dutch Elm Disease, with various interventions having been developed theoretically or in laboratory conditions (e.g. Scheffer & Strobel 2020, Chavez et al. 2015, Marciulyniene et al. 2017, Halecker et al. 2020).

3.6.1.2 Co-Benefits and Trade-offs

Better tree health could contribute to landscape quality and carbon sequestration.

3.6.1.3 Magnitude

Tree health will have little effect at landscape scales, but effects could be large and long-term under extreme pressure from issues such as Ash Dieback. Fundamentally, however, this is unpredictable as effects will depend on the scale of any new disease threats.

3.6.1.4 Timescale

Monitoring needs to be indefinite but negative effects of disease could be apparent within a few years; positive effects of prevention are very hard to detect.

3.6.1.5 Spatial Issues

Monitoring will be most needed and effective where disease threats are prevalent, if they vary spatially – so potentially closer to ports or to existing, known disease hotspots, for example.

3.6.1.6 Displacement

N/A

3.6.1.7 Maintenance and Longevity

Monitoring needs ongoing, indefinite effort. Long-term effectiveness depends on successful mitigation measures: it may not be possible to control disease spread, regardless of monitoring.

3.6.1.8 Climate Adaptation or Mitigation

Better tree health could contribute to landscape connectivity and carbon sequestration.

3.6.1.9 Climate Factors / Constraints

Disease threats will vary over time and novel threats are likely to arise with climate change, so monitoring approaches (target tree species and mechanisms to detect disease agents) may need to be adapted to keep pace.

3.6.1.10 Benefits and Trade-offs to Farmer/Land-manager

None

3.6.1.11 Uptake

Effective monitoring for rare and cryptic events may be difficult to sustain in the long term if they are not recorded.

3.6.1.12 Other Notes

None

3.7 BUNDLE: NATURAL REGENERATION

This broad area of action can be interpreted as 'rewilding' at different scales – gross habitat change using natural processes to return land to something like natural habitat, with native wildlife communities. Where goals are clearly defined and active interventions are used more, this set of interventions increasingly overlaps with those in Section 3.8 Restoration, Management and Enhancement.

3.7.1 ETPW-171 Allow natural regeneration and extension of existing habitat (e.g. hedgerows, scrub, rough grassland) and ETPW-171 Protect natural regeneration eg (through scrub management, protective fencing, invisible fencing)

This action appears to refer to more-or-less passive rewilding, probably at small scales, but promoting desirable species and removing undesirables, managing habitat e.g. by mowing, grazing or fencing. The definition here is somewhat contradictory, as manipulated 'natural' regeneration is then clearly not natural, but neither are invasives that are removed through scrub management.

3.7.1.1 Causality

Natural regeneration will deliver appropriate habitat for native species, although its effectiveness depends on the specific goals as well as the actions that are taken: wild habitat will spread through natural succession in all contexts, but the precise vegetation content and structure could be manipulated in various ways and the success of the actions taken can only be judged against their specific goals. Therefore, clear goals in terms of a target vegetation assemblage or community structure are essential. Further, some manipulation of natural processes is likely to be needed to deliver the goal, for example to control grazing/browsing pressure, to prevent the establishment of native species or to reduce soil fertility. There does not appear to be a large evidence base for this in the English context.

Fencing allows regeneration in the face of deer browsing pressure, but does not necessarily deliver recovery of the dominant species in the target assemblage (Perrin et al. 2006), but also works well in other contexts (Garbutt & Wouters 2008). Effective control of dense Pteridium stands has been found to be necessary to allow ‘natural’ regeneration of oak, for example (Humphrey & Swaine 1997).

Abandoned farmland will go through vegetation succession, eventually reaching the climax vegetation for the location and context, as modified by factors such as grazing. Hence, any ambition to create habitat at an intermediate stage of succession will require ongoing management. Alternatively, targets or outcomes can be framed in terms of the ongoing process instead of the nature of the ultimate habitat (Hughes et al. 2011).

3.7.1.2 Co-Benefits and Trade-offs

There will be an inevitable loss of productive land, while new natural habitat will be a source of both species providing ecosystem services and those with negative effects.
Carbon sequestration – or avoidance of loss.
Flood prevention (better infiltration than farmland).

3.7.1.3 Magnitude

Effects are likely to be large locally but small at a landscape scale because uptake would not be large, in practice. Even large-scale current rewilding schemes still only have a small footprint at the landscape scale and are unlikely to have a significant impact on national populations.

3.7.1.4 Timescale

Long-term responses of vegetation are likely to take decades, although more active management (e.g. tree-planting) could increase response speed.

3.7.1.5 Spatial Issues

This will probably occur most on land of low agricultural value, so found in particular regions where the need for restoring native biodiversity may not be the greatest; it is likely to be unpopular in areas of intensive agriculture, where it might be needed the most.
Success and effects may well depend on spatial context, patch size and connectivity, i.e. how the habitat created is able to support within the patch or across the landscape in which the patch lies, as well as the ease of natural colonisation by desirable native species (and problem species with negative effects).

3.7.1.6 Displacement

This will probably be used most on land of low agricultural value, so leading to no significant issues with displacement in practice.

3.7.1.7 Maintenance and Longevity

Long-term, ongoing maintenance will be needed at least until habitat is mature, assuming that some target vegetation type is intended.

3.7.1.8 Climate Adaptation or Mitigation

Depending on the habitat, there is clear potential value as a local carbon sink, or at least a cessation of carbon losses from production.

3.7.1.9 Climate Factors / Constraints

Vegetation may need more active management to proof the developing communities against future climates, such as assisted colonisation.

3.7.1.10 Benefits and Trade-offs to Farmer/Land-manager

There will be an inevitable loss of productive land, while new natural habitat will be a source of both species providing ecosystem services and those with negative effects. Naturally regenerating field margins supported good pollinator numbers, but via pernicious weeds, unlike sown margins containing more desirable plant species (Pywell et al. 2005).

3.7.1.11 Uptake

Rewilding is not popular with farmers and may also be seen as a negative by changing cultural landscapes.

3.7.1.12 Other Notes

None

3.8 Bundle: Restoration, management and enhancement

This section is assumed to describe actions for active habitat change, as opposed to allowing passive rewilding (Section 3.7). The distinction is starting a conversion process with a clear goal in mind in terms of the habitat to be created and then the application of specific interventions to create that habitat.

3.8.1 EBHE-007 Create/ restore/ manage traditional field boundaries (eg dry stone walls, earth banks, stone faced earth banks, Cornish hedges) and EBHE-019 Create/ maintain appropriate boundary features alongside rights of way such as hedges, bird watching cover and dry stone walls

This family of actions is fundamentally concerned with preserving landscape character by ensuring that new and restored boundary features are consistent with others in the landscape. The specifics of the management will therefore vary considerably between locations and there will not be large changes in the resources provided for biodiversity.

3.8.1.1 Causality

Reviewing this action and considering effects as a single line of evidence is problematic since all forms pof boundary are different and their ecological effects will inevitably differ. Walls can be significant in their own right as habitats and corridors for amphibians, bees, butterflies, birds, insects etc. (Wright 2016, Powell et al. 2019). They can also be a determinant of wheatear density as a source of nest sites (Woodhouse et al. 2005), but there has been little research on them, especially in the UK (Collier 2013). Grassy boundary features like banks provide resources for many butterfly (Sparks & Parish 1995) and bird (Vickery, Carter& Fuller 2002) species. However, again there has been little research into these features being added as agri-environment interventions, specifically. Hedges are reviewed in detail in under action **ECCM-025EM**; it would be expected that their effects would be the same as in the present context as ‘appropriate’ or ‘traditional’.

3.8.1.2 Co-Benefits and Trade-offs

Landscape character

3.8.1.3 Magnitude

Effect sizes are likely to be small because these are not major features of the landscape, but they could be significant at small scales, and hedgerow effects could be large if management is conducted over large scales (see **ECCM-025EM**), although the evidence for hedgerow management effects on birds shows only small impacts at most. Effects will also be variable between the boundary types here.

3.8.1.4 Timescale

Hedgerow management benefits should begin to appear within two or three years, although restoration of highly degraded hedges is likely to take longer as planted shrubs mature. Responses to the other boundary types may be quicker.

3.8.1.5 Spatial Issues

The habitat context of boundaries is critical to their effects, involving both other AES measures and broader habitat types, presence and connectivity.

3.8.1.6 Displacement

Enhanced (higher, wider) hedges may have negative effects the productivity of very adjacent land by shading, but such parts of fields are likely not to be highly productive anyway. In general, boundary management is likely to have minimal displacement effect.

3.8.1.7 Maintenance and Longevity

Hedgerow management needs to be regular and indefinite, but ideally not annual, so there are benefits relative to a standard annual management scheme. Other boundary types require relatively little maintenance in the long term.

3.8.1.8 Climate Adaptation or Mitigation

Hedgerows as agents of connectivity could play a role in climate adaptation.

3.8.1.9 Climate Factors / Constraints

None.

3.8.1.10 Benefits and Trade-offs to Farmer/Land-manager

Hedges, especially thick or tall ones, can have negative effects on farm operations, making vehicular access more difficult and shading crops. Ideal hedgerow management dates may fall at times of the year when it is not convenient to farmers, e.g. when fields are wet, so accessing boundary vegetation for cutting may mean damaging the ground. This may limit uptake where soils are heavy or rainfall is high. The other boundary types should be neutral.

3.8.1.11 Uptake

Bigger, higher, wider hedgerows may appear less neat or make farm operations more awkward, making farmers less willing to take up such management. Ideal hedgerow management dates may fall at times of the year when it is not convenient to farmers, e.g. when fields are wet, so accessing boundary vegetation for cutting may mean damaging the ground. This may limit uptake where soils are heavy or rainfall is high.

Promoting the quality of rights-of-way may be unpopular with farmers who do not want to encourage access to their land.

3.8.1.12 Other Notes

None

3.8.2 ETPW-217 Create areas of scrubby flower-rich grassland

Flower-rich grassland can be created from temporary grassland or arable land, but the land needs to be low in fertility. In grassland, the method can be oversowing yellow rattle, spreading green hay, sowing seed or plug planting. Arable land needs to be prepared as a firm, fine, level and weed-free seedbed, into which wild flower and grass seed can be sown. It may also be possible to do this by natural regeneration, if the seed bank is rich and/or there are local sources of desirable species. A scrubby element is likely to develop via natural succession, or could be facilitated with plugs of desirable species. It is likely that scrub will need ongoing management to be maintained at a desired level.

3.8.2.1 Causality

There appears to be no evidence relating to scrubby, flower-rich grassland per se. Species-rich grassland is reviewed under action **ECPW-022** and the same evidence is relevant here. The key difference will be the addition of scrub, either directly or by arresting succession at a later stage than in 'pure' grassland. The amount of scrub added or allowed to persist could be critical to value for various species. In most landscapes, this habitat would provide a valuable addition as habitat for plants and invertebrates, with consequent benefits for mammals and birds given sufficient scale of implementation. If the habitat is created from intensive farmland, it should always have a net positive effect on biodiversity, with very few negative effects.

3.8.2.2 Co-Benefits and Trade-offs

Carbon benefit from reduction in cultivation/grazing

3.8.2.3 Magnitude

Like other species-rich grassland, this could have a significant benefit for a range of groups, especially locally.

3.8.2.4 Timescale

See **ECPW-022**.

3.8.2.5 Spatial Issues

See **ECPW-022**.

3.8.2.6 Displacement

Limited effects as on low-fertility land.

3.8.2.7 Maintenance and Longevity

Will need ongoing mowing, light grazing and/or scrub control.

3.8.2.8 Climate Adaptation or Mitigation

See **ECPW-022**.

3.8.2.9 Climate Factors / Constraints

See **ECPW-022**.

3.8.2.9.1 Benefits and Trade-offs to Farmer/Land-manager

See **ECPW-022**.

3.8.2.10 Uptake

A commitment to ongoing scrub control may be unattractive to farmers and affect uptake.

3.8.2.11 Other Notes

None

3.8.3 ECCM-025EM enhance/ manage hedgerows & ECCM-080EM enhance/ manage hedgerows around point-source polluters

Hedgerows are traditional field boundary structures in many UK landscapes, especially in the lowlands. However, they are often not actively maintained and hence in poor condition, especially in arable farmland, where they have little agricultural function, but also in pastoral systems, where fences have taken over the primary role in stock control. For hedgerows to retain their agricultural function and structural integrity, they need to be managed regularly by trimming, but the timing and frequency of trimming has important effects on the vegetation height and density, on flower and berry densities, and on risk of disturbance to wildlife. Management to enhance hedgerows can consist of restoration to recreate contiguous linear scrub habitat by planting shrubs and 'laying' them to knit the plants together. Enhanced management then involves cutting regimes to deliver desirable vegetation characteristics.

Hedgerows vary in height, width, plant species composition, the presence of trees and ditches, and degree of connectivity with other woody habitat.

3.8.3.1 Causality

Hedgerows provide unique networks of semi-natural habitat in production landscapes, both supporting communities in their own right and providing the resources that allow various species also to use in-field, cropped or grazed, field habitats. The resources they provide include cover, shelter, microclimate, refugia from crop spray effects, corridors for movement (connectivity), plant or animal food, seeds, nectar, pollen and nest sites. They may have interacting effects with adjacent habitats, such as AES field margins or agricultural crops. Management to restore to enhance hedgerows should increase these effects.

Landscape-scale studies of changes in hedgerows due to management have been difficult because of a lack of detailed, large-scale data on hedgerow locations and structure, a limitation that is now easing with the growing availability of remote-sensed data such as LiDAR. However, hedgerows and field margin vegetation affect the richness and abundance of flora, invertebrates and birds (Boatman (ed) 1994, De Snoo, 1999 and Hinsley and Bellamy, 2000), and there is good evidence that the structure and form of a hedgerow and its management, in terms of differences in width, height, fenced buffer strips, frequency of cutting, etc., has a large effect on biodiversity (Haddaway et al. 2018).

Heterogeneity in hedgerow structural condition is important because no single set of hedgerow characteristics have been found to benefit all taxa and, if uniform hedgerow management is overprescribed, some species are likely to be adversely affected by a loss of suitable habitat or resource decline (Graham et al., 2018). A report for Defra, focused on hedgerow priority species (former BAP species) and those listed as Biodiversity 2020 Farmland Indicators, presents evidence of the importance of the inter-relationship of the five structural components of hedges (trees, shrubs, hedge base, field margins and ditches). Overall, of the 107 species studied, the majority (65%) are dependent on more than one hedge component, and over a third of them (35%) are dependent on three or more components (Wolton et al, 2013).

Hinsley & Bellamy (2000) reviewed the value of hedgerows as bird habitat in lowland-farming landscapes to provide a background against which decisions concerning hedgerow management might be evaluated. The two most important factors positively associated with species richness and abundance of breeding birds in hedgerows were hedge size (height/width/volume) and the presence/abundance of trees. The provision of cover and the botanical and structural complexity of the vegetation are also important. However, large hedges do not suit all species; birds tend to prefer hedgerow types which most closely resemble their usual non-hedgerow breeding habitat. The value of hedgerows to birds can be increased by combining them with other features such as headlands (for game birds), verges, wildflower strips, game and wild-bird cover and well-vegetated banks and ditches. The presence of well-grown, dead or decaying trees is beneficial to many species, providing nest holes, foraging sites and perches. Increasing the structural complexity of a hedgerow and its associated habitat may also reduce the incidence of predation. Hedgerows also provide physical shelter and roost sites and are an important source of winter food supplies, especially berries and other fruits. Some bird species, usually those whose primary habitat is woodland, live mainly within the hedgerow itself, whereas others are more dependent on the surrounding landscape to a greater or lesser extent. However, even the presence of woodland bird species is influenced by the availability and characteristics of alternative habitats in the surroundings and therefore hedgerows and their bird populations do not function as isolated patches. As linear landscape elements, hedgerows also provide safe cover for both local and larger-scale movements and may facilitate access to resources or habitat which might otherwise be too risky or too remote for birds to use or colonise. Overall, it is clear that a variety of hedge management practices are likely to be required to maximise the resources for birds and the resultant diversity and abundance, and that the habitat context of hedgerows is critical to their effects, involving both other AES measures and broader habitat types, presence and connectivity.

Hedgerow management under agri-environment schemes is associated with greater use by hedgerow bird species (Davey et al. 2010, Redhead et al. 2013) but there is only limited evidence for benefits to species' population growth rates at a national scale (Baker et al. 2012) probably because this management benefits breeding productivity, but most species are limited by over-winter survival. It is inherently hard to collect empirical evidence (and thus parameterise models) of causality about use of hedgerows by species as corridors (Davies and Pullin 2007).

There is limited evidence of the impact of hedgerow management under AES on birds at the field scale. Results from UK studies have shown a mixed relationship between hedgerow management and bird populations. The more sophisticated analyses found different relationships for hedgerow management and possibly changing relationships over time, but with no clear patterns for positive influences across species (Pringle et al. 2018). However, there was a clear positive relationship between hedgerow management and population growth rate within Tir Gofal in Wales, with five of eleven species tested showing significant associations with hedgerow management (Dunnock, Greenfinch, House sparrow, Song Thrush and Linnet), all of which were positive (Siriwardena & Dadam 2015), and showing that population effects can be detected under the correct sampling regime. Of course, many farmland passerines may not be limited by breeding success or nest site availability, in which case a clear population response to hedgerow management is not necessarily expected.

Appropriate hedgerow management may be critical to allow early flowering woody hedgerow species such as blackthorn to fill the gap for resources for pollinators in spring (Staley et al. 2012, Dicks et al. 2015). Hedgerow management may provide nesting resources for pollinators (Dicks et al. 2015). Just one study found an increase in abundance of nest searching queen bumblebees associated with hedgerows, species rich grassland and grass margins / beetle banks in Scotland (Lye et al. 2009). Staley *et al.* (2014) found that hedgerows managed under the ELS option EB1 (cut once every two years in early autumn) did not have significantly higher numbers of Brown hairstreak eggs than a control hedge, but one variant of the HLS and ELS option EB3 (cut at most once every three years in early autumn) had significantly greater numbers of eggs than control hedges. Whether hedgerow management has an effect on moths is dependent on the AES option under consideration, interactions with management of other habitat features and moth functional grouping. The presence of hedgerow trees in fields increased the abundance of macro-moths that feed on grass or herbs, but had no effect on the abundance of those species that feed on shrubs or trees (Merckx et al. 2010), possibly because grass- or herb-feeding species have lower mobility and thus are more dependent on the shelter provided by isolated trees (Merckx et al. 2010). Merckx et al. (2012) found positive effect of hedgerow tree presence on total moth abundance, but not species richness. Further, Merckx et al. (2009) found that positive effects from hedgerow trees on the abundance of large moth species were not significant on their own, but were when applied in combination with 6m grass margins (ELS option EE3), and in landscapes where AES option uptake was actively encouraged. Staley et al. (2016) showed no difference in abundance or species richness of moths on non-AES hedgerows and those cut once every two years in the autumn (ELS option EB2), whereas hedges cut every three years in winter (which fall under ELS option EB3) had significantly higher abundance.

Studies of Dormouse responses to hedgerow or boundary management that have shown spatial relationships (Bright & MacPherson 2002), although there is no clear evidence of effects on population change.

Species such as skylark and lapwing that respond negatively to hedgerow presence may respond more negatively to enhanced, taller, thicker hedges, but this effect is likely to be marginal: the landscapes involved will already be either broadly suitable or unsuitable.

3.8.3.2 Co-Benefits and Trade-offs

None.

3.8.3.3 Magnitude

This will be highly variable between species and taxonomic groups, and dependent on what precise action is taken to improve biodiversity management of hedgerows. Hedgerow effects on particularly hedge-associated or hedge-avoiding species could be large if management is conducted over large scales.

3.8.3.4 Timescale

Hedgerow management benefits should begin to appear within two or three years, although restoration of highly degraded hedges is likely to take longer as planted shrubs mature.

3.8.3.5 Spatial Issues

The habitat context of hedgerows is critical to their effects, involving both other AES measures and broader habitat types, presence and connectivity.

3.8.3.6 Displacement

Enhanced (higher, wider) hedges may have negative effects the productivity of very adjacent land by shading, but such parts of fields are likely not to be highly productive anyway. In general, boundary management is likely to have minimal displacement effect.

3.8.3.7 Maintenance and Longevity

Hedgerow management needs to be regular and indefinite, but ideally not annual, so there are benefits relative to a standard annual management scheme.

3.8.3.8 Climate Adaptation or Mitigation

Hedgerows as agents of connectivity could play a role in climate adaptation, and enhanced hedgerows could play more of a role, but the uncertainty about their true role in this regard (see above) should be taken into account.

3.8.3.9 Climate Factors / Constraints

None.

3.8.3.10 Benefits and Trade-offs to Farmer/Land-manager

Hedges, especially thick or tall ones, can have negative effects on farm operations, making vehicular access more difficult and shading crops. Ideal hedgerow management dates may fall at times of the year when it is not convenient to farmers, e.g. when fields are wet, so accessing boundary vegetation for cutting may mean damaging the ground. This may limit uptake where soils are heavy or rainfall is high.

3.8.3.11 Uptake

Bigger, higher, wider hedgerows may appear less neat or make farm operations more awkward, making farmers less willing to take up such management. Ideal hedgerow management dates may fall at times of the year when it is not convenient to farmers, e.g. when fields are wet, so accessing boundary vegetation for cutting may mean damaging the ground. This may limit uptake where soils are heavy or rainfall is high.

3.8.3.12 Other Notes

None

3.8.4 TPW-038 Create/ manage/ enhance buffer strips and ECPW-042 Create/ enhance/ manage riparian buffer strips

Ecologically, this is equivalent to the actions considered in 3.1.2 (**ETPW-271**), so the action is not reviewed further here. Benefits for rivers are possible, but not in scope at this stage.

3.8.5 EBHE-192 Manage existing in-field trees situated within areas of cultivated land by reversion to permanent pasture to beyond extent of tree canopy to protect tree roots from cultivation and compaction

In-field trees are a potentially important source of habitat heterogeneity, connectivity and resources such as nest sites in farmed landscapes, but are vulnerable to agricultural operations such as ploughing and vehicle traffic compacting the soil, with deleterious effects on root systems and tree health.

3.8.5.1 Causality

There is a lack of specific research on the effects of management of in-field trees. The value of individual trees is considered under **ECPW-156** Plant, enhance or manage trees and shrubs and **ECCM-056** Manage veteran and ancient trees; the specific feature here is that the trees face some different threats and pressures from being within farmed areas, as opposed to boundary features, and that the trees are not necessarily 'veteran' or 'ancient'. The specific actions described for protection are a subset of those that could be applied under **ECCM-056**, but the effect would still be to prevent loss or to extend the life of existing trees. Hence, direct evidence would involve the benefit of maintenance instead of loss, which would be difficult to demonstrate. Benefits of younger trees will be similar to those of veteran trees, but probably smaller in extent.

3.8.5.2 Co-Benefits and Trade-offs

[TOCB Report-3-6 *Carbon* **EBHE-192**] Compaction and root damage are deleterious to woody vegetation, and can lead to physiological disfunction that can lead to symptoms of drought and nutrient deficits if severe (Kozlowski, 1999). Silvo-pasture trees can also store large amounts of carbon. A study by Upson (2016) found that a 14-year-old silvo-pastoral system with a combination of trees and grassland stored about 5 per cent more carbon (a combination of above ground and soil carbon) than the equivalent separate areas of woodland and pasture, putting this down, at least in part, to the greater size of the silvo- pastoral trees.

Whilst there is not empirical evidence assessing the effects of avoiding compaction on rates of carbon sequestration and storage in trees specifically, there is a robust logic chain suggesting that this will potentially increase the ability of trees to sequester carbon and preserve existing carbon stocks, particularly in old trees.

Food and fibre production	Area under production or yield and outside of ELM	**N**
Global, regional & local climate regulation	Above ground carbon sequestration	**L***
	Below ground carbon sequestration	**L***

3.8.5.3 Magnitude

See **ECCM-056**.

3.8.5.4 Timescale

See **ECCM-056**.

3.8.5.5 Spatial Issues

See **ECCM-056**.

3.8.5.6 Displacement

See **ECCM-056**.

3.8.5.7 Maintenance and Longevity

See **ECCM-056**.

3.8.5.8 Climate Adaptation or Mitigation

See **ECCM-056**.

3.8.5.9 Climate Factors / Constraints

See **ECCM-056**.

3.8.5.10 Benefits and Trade-offs to Farmer/Land-manager

See **ECCM-056**.

3.8.5.11 Uptake

See **ECCM-056**.

3.8.5.12 Other Notes

None

3.8.6 ECCM-056 Manage veteran and ancient trees

Veteran trees clearly occur in woodland, but this action is interpreted as referring to those in farmland contexts, i.e. mostly in field boundaries. These trees are potentially threatened by agricultural activity and may be more vulnerable to environmental pressures such as storm damage than those in woodland. Recruitment of new trees in a field boundary context may also be rare, so the protection of existing trees is particularly important to maintain their ecological function.

3.8.6.1 Causality

Evidence for this action is captured under **ECPW-156** plant, enhance or manage trees and shrubs. Veteran trees have additional benefits for a wide range of taxa, particularly invertebrates, but also nest holes for various bird species, for example (Horák 2017, Siitonen & Ranius 2015, Wetherbee et al. 2020, Nowak 2004). This particularly involves specialist invertebrates (Sverdrup-Thygeson et al. 2017). Management of veteran trees per se has not been tested as an intervention.

3.8.6.2 Co-Benefits and Trade-offs

No assessment.

3.8.6.3 Magnitude

Effects may be large locally but those of individual trees will be limited in scope. Networks of veteran trees could have landscape-level effects and be very significant for particular invertebrate groups, in particular.

3.8.6.4 Timescale

Benefits of these trees already exist; this management is to maintain them.

3.8.6.5 Spatial Issues

Benefits of veteran trees will be greater where they are close together or connected in the landscape.

3.8.6.6 Displacement

N/A

3.8.6.7 Maintenance and Longevity

Veteran trees need to be protected and managed indefinitely, so this is an ongoing, not time-limited, action.

3.8.6.8 Climate Adaptation or Mitigation

Veteran trees can provide important contributions to landscape connectivity for particular groups, such as beetles.

3.8.6.9 Climate Factors / Constraints

Protecting trees under a changing climate could be increasingly problematic and could become impossible if the climate changes too much.

3.8.6.10 Benefits and Trade-offs to Farmer/Land-manager

N/A

3.8.6.11 Uptake

Farmers might prefer to allow inconveniently placed trees to die rather than to protect them, which could affect uptake of specific protection measures.

3.8.6.12 Other Notes

None

3.9 BUNDLE: SIGNPOSTING, INFORMATION, FACILITIES AND EVENTS

In general, these actions will not have direct environmental or biodiversity effects, although informing, educating and directing the public could enhance protection locally and encourage political will to support management activities more broadly. However, some forms of action could have direct effects on environmental outcomes.

3.9.1 EBHE-058 Create small-scale cultivation opportunities and EBHE-059 Create/ maintain sites and small scale infrastructure for community therapeutic horticulture or food growing

This is interpreted as referring to allotments and community farms, wherein new cultivations can be made close to human habitation. The primary driver here is for human engagement with food production and community/health benefits, but habitat change with biodiversity effects is likely and the new landscape would probably have a finer spatial grain than is usual in farmland, reflecting the interests of multiple managers.

3.9.1.1 Causality

Effects here will inevitably depend on the functional definition of 'small-scale': assume it relates to sites that are too small to qualify for BPS. Little evidence collected as these are understudied. Effects of management will probably be small effects by definition, because the areas involved will be small, although there could be local, significant effects. This form of habitat change has not been investigated or monitored to date, and there are opportunities to refine the management that takes place. Actual land-use may change little, e.g. arable land remaining arable, but land-use heterogeneity is likely to increase, even if only in terms of arable/horticultural crop diversity.

3.9.1.2 Co-Benefits and Trade-offs

Not assessed

3.9.1.3 Magnitude

This can only have small-scale, local benefits, but may have disproportionate effects on farmland biodiversity as experienced and engaged with by farmers.

3.9.1.4 Timescale

Effects are likely to occur within a year or two.

3.9.1.5 Spatial Issues

Effects will be greater where there is a larger contrast in land-use following the introduction of small-scale farming in a location.

3.9.1.6 Displacement

This is likely to be minimal as production land will still be used for production, although it is possible that a small area of intensive agriculture will be displaced.

3.9.1.7 Maintenance and Longevity

Creating small-scale agriculture opportunities would be a single event, but long-term support may be required to make production there economical.

3.9.1.8 Climate Adaptation or Mitigation

N/A

3.9.1.9 Climate Factors / Constraints

N/A

3.9.1.10 Benefits and Trade-offs to Farmer/Land-manager

None are clear.

3.9.1.11 Uptake

Availability of land may be a key restriction: suitable agricultural land needs to be available in a target area and landowners may not be willing to hand over or to sell land. Initial uptake and long-term sustainability will depend upon public interest to engage with small-scale agriculture in the area. This suggests that locations should be informed by prior public engagement.

3.9.1.12 Other Notes

None

3.10 Bundle: Specific wildlife targeted actions

This section covers a diverse set of actions involving management specifically for wildlife that is more-or-less independent of agricultural processes: some involve interventions entirely within the semi-natural or marginal habitats within a farm, while others involve taking land out of production entirely, permanently or temporarily.

3.10.1 EBHE-182 Create and use a wildlife management plan

A management plan is a description of the short-term objectives and long-term goals that will be met by manipulation of habitat, wildlife populations, and people and how these objectives and goals will be reached (Anderson et al. 2002). Management planning is the intellectual or 'thinking' component of the conservation management process (Alexander 2020).

3.10.1.1 Causality

A plan alone does not deliver any outcomes; other actions need to follow. Hence, while plans are important, especially for complex situations where multiple actions are proposed to meet multiple specific targets, they are fundamentally concerned with coordinating other actions. However, plans also have to be well-designed, so that land managers and farmers can relate to them as relevant and meaningful on the ground, in a practical sense. If plans are formulated poorly, they will not be followed (Alexander 2020). Alexander (2020) presents a good guide to the application of management plan in a conservation context; they are likely to be still more important, and more important to be practicable, when conservation motivations are combined with those involved in agricultural production. In an AES context, wildlife-specific planning can be integrated within analogues of the Farm Environment Plan (FEP) that was a feature of Higher-Level Stewardship (Smallshire et al. 2004, Martin 2012). The most successful planning approaches have involved one-to-one consultations with advisers (Winter et al. 2000) and NE staff report informally that regions with dedicated, coordinated advice, such as Farmland Bird Initiative areas, have improved delivery of options that are normally less popular with farmers and better coordination at the landscape scale, but it is very difficult to prove biodiversity benefits because of the lack of counterfactuals for such regions. A key benefit of informed planning is that packages of interventions for an individual farm can be tailored to the needs of local wildlife and to filling the resource gaps to enable a wide range of taxa to benefit (Winspear et al. 2010), rather than for actions to be determined by agronomic factors or farmer preference. However, a weakness is that frequently combining sets of interventions can compromise the identification of the effects of individual actions, as they are then confounded.

3.10.1.2 Co-Benefits and Trade-offs

Wildlife management planning can be integrated with plans for other environmental outcomes and hence facilitate complementary results.

3.10.1.3 Magnitude

This will be plan-, taxon- and context-specific.

3.10.1.4 Timescale

This will be plan-, taxon- and context-specific.

3.10.1.5 Spatial Issues

This will be plan-, taxon- and context-specific.

3.10.1.6 Displacement

This will be plan-, taxon- and context-specific.

3.10.1.7 Maintenance and Longevity

This will be plan-, taxon- and context-specific, but plan commitments are likely to be indefinite, or at least long-term.

3.10.1.8 Climate Adaptation or Mitigation

Relevant measures could be built into plans.

3.10.1.9 Climate Factors / Constraints

Plans should consider climate-dependency to have long-term relevance and value.

3.10.1.10 Benefits and Trade-offs to Farmer/Land-manager

These will be plan-, taxon- and context-specific, but plans would allow best integration with farm management to minimise problems and to avoid barriers.

3.10.1.11 Uptake

Farmers may not be convinced of the value of plans that may conflict with their current practices and future plans.

3.10.1.12 Other Notes

None

3.10.2 EBHE-302 **Install/ maintain bird and insect houses and** ETPW-200y **Provide nesting and roosting sites (e.g. nesting boxes, bat boxes)**

This action covers a range of specific interventions involving the installation of physical infrastructure to provide shelter or nest sites for various different taxa.

3.10.2.1 Causality

Mostly minimal as few boxes will be installed in any one location, but could be important if the resource involved is limiting for the presence or abundance of key species. This is perhaps most likely to lead to a detectable effect for species that are at low densities, so the addition of resources to support one pair or colony would affect presence at site level significantly. The Conservation Evidence project[3] has summarised the evidence for a range of actions that fall within these broad categories, finding that boxes for birds are either 'beneficial', 'likely to be beneficial' or have 'unknown effectiveness due to limited evidence', depending on the specific sub-category. The 'unknown' cases involve woodpeckers (in North America), owls, swifts, and the specific actions of providing boxes (i) to reduce inter-species competition (in North America) and (ii) translocating them to reduce predation (by Pine Martens in continental Europe). Bird nest boxes can (a) provide nesting habitat where natural holes are otherwise absent or (b) provide more secure nesting habitat where nests are vulnerable to predation or weather effects, but only benefit cavity-nesting species, so are not a solution for most birds of conservation concern in farmland. Success in the Conservation Evidence scheme has generally been assessed in terms of box occupancy, which does not necessarily imply a population benefit, but boxes are at likely at least to be neutral for the target population.

Nest boxes for bees have had varying levels of possible benefit (occupancy), linked to some extent to the specific design of the box, although there is no clear evidence for population benefits (Conservation Evidence[4]). Conservation Evidence[5] also found that boxes for roosting bats are 'likely to be beneficial', but again all studies have involved box use, rather than population benefits.

Maintaining box structures is clearly likely to maintain their functions, while cleaning them regularly can reduce parasite loads and disease risks. Some studies (although not none in the UK) have found that various species select cleaned boxes, but there have been no studies of the demographic or population effect of this choice (Conservation Evidence[6]).

3.10.2.2 Co-Benefits and Trade-offs

There is some evidence that providing nest boxes for key predatory species can reduce the abundance of pests in orchard systems (Shave et al. 2018, García et al. 2021).

3.10.2.3 Magnitude

This will vary case by case and be entirely dependent on the extent to which boxes limit the specific population. Note that benefits beyond the target species, i.e. at an ecosystem level, are unlikely.

3.10.2.4 Timescale

Boxes can be occupied within the season in which they are provided, or can take some years to be found, depending on the limiting factors on local abundance. Demographic and population effects are likely to take several years to be detectable.

3.10.2.5 Spatial Issues

Boxes need to be placed where the target species has source populations that can find them.

3.10.2.6 Displacement

N/A

3 https://www.conservationevidence.com/data/index?terms=nest%20box&yt0=
4 https://www.conservationevidence.com/actions/48, https://www.conservationevidence.com/actions/47 , https://www.conservationevidence.com/actions/80
5 https://www.conservationevidence.com/actions/1024
6 https://www.conservationevidence.com/actions/499

3.10.2.7 Maintenance and Longevity

Boxes need to be maintained and cleaned, ideally annually, and will need to be replaced when damaged by weather events or non-target animals like squirrels.

3.10.2.8 Climate Adaptation or Mitigation

Boxes may play a role in allowing species to move to new habitats (by natural dispersal or translocation) as a climate change adaptation measure.

3.10.2.9 Climate Factors / Constraints

None.

3.10.2.10 Benefits and Trade-offs to Farmer/Land-manager

These are measures with very low impact on farm operations, so achieve a conservation outcome with minimal inconvenience, although they will only ever benefit a subset of species of concern.

3.10.2.11 Uptake

N/A

3.10.2.12 Other Notes

None.

3.10.3 ECCA-034 Create, enhance, manage natural refugia

'Refugia' is a term that is used in several contexts, including locations providing animals with shelter from challenging conditions or agricultural operations, sites where plants can grow and set seed indefinitely, and hence continually colonise agricultural habitats nearby as they rotate through phases of suitability, large-scale habitat environments where climate change effects are buffered, and habitat patch sources of beneficial invertebrates. Such habitats therefore aim to provide resilience to challenges for aspects biodiversity and certain ecosystem services. However, the range of specific, potential actions and species/contexts to which this could be applied is very large; only a few specific instances are considered below. In some cases, 'refugia' are provided as one of the functions of other interventions, such as field margin buffers.

3.10.3.1 Causality

Field margins provide refugia for species such as carabid beetles that, in turn, may perform a pest control service in crops (Thomas & Marshall 1999, Dennis & La Fry 1992), for butterfly species of conservation value (Pywell et al. 2004) and for some but not all of the flora locally associated with low-fertility grassland (Smart et al. 2002). Similarly, they can support reservoir populations of arable plants long after they have been lost from field centres due to intensive arable management (Fried et al. 2009). However, they can also provide refugia for ticks *Ixodes ricinus*, especially near woodland, and hence have a deleterious effect on humans and livestock (Medlock et al. 2020). Margins can also provide refugia, cover or corridors for predators of species of conservation interest, potentially negating the otherwise-positive effects of in-field actions such as skylark plots (Morris & Gilroy 2008).

Modelling based on national plant and invertebrate species record data for multiple taxa suggests that the English protected area network generally features environmental characteristics that should allow the potential of the landscapes to act as refugia and has helped historical population persistence (Suggitt et al. 2014). However, relationships were less clear at smaller spatial scales, where microclimatic challenges to species will be more variable, so making specific recommendations is difficult (Suggitt et al. 2014).

Nevertheless, the general message is that protection of currently values, diverse, semi-natural habitats will deliver the refugium function.

Artificial refugia from inclement weather perform both a resource provision and a monitoring function for reptiles (e.g. Sewell et al. 2013). However, it is unclear how important such resources are for population persistence.

Overall, the range of interventions, targets and contexts here is too great to provide any kind of clear, integrated score.

3.10.3.2 Co-Benefits and Trade-offs

There is a possible pest control service from some types of refugia.

3.10.3.3 Magnitude

This will vary case by case and be entirely dependent on the extent to which refugia limit the local abundance of the specific population concerned.

3.10.3.4 Timescale

This will vary case by case.

3.10.3.5 Spatial Issues

Clearly refugia need to be provided where they are currently absent or present in quantities that are too low to support the needs of target populations.

3.10.3.6 Displacement

N/A

3.10.3.7 Maintenance and Longevity

Refugia are highly varied in form; some may need cleaning, renewal or replacement from time to time, perhaps annually.

3.10.3.8 Climate Adaptation or Mitigation

Refugia may allow populations to persist where the climate is becoming less suitable, or to colonise new habitat areas where the climate is expected to become more suitable.

3.10.3.9 Climate Factors / Constraints

N/A

3.10.3.10 Benefits and Trade-offs to Farmer/Land-manager

These are measures with very low impact on farm operations, so achieve a conservation outcome with minimal inconvenience, although they will only ever benefit a subset of species of concern.

3.10.3.11 Uptake

N/A

3.10.3.12 Other Notes

None

3.10.4 EHAZ-054 Create/ enhance/ manage scrapes

Scrapes are shallow, perhaps seasonal, pools created in open field or marsh contexts to provide habitat for aquatic and marginal birds, invertebrates or plants. They are particularly aimed at breeding wading birds, providing key feeding and refuge habitat.

3.10.4.1 Causality

Dicks et al. (2017) reported that five studies (including a replicated, controlled, paired trial) from Sweden and the UK found creating scrapes and pools provided habitat for birds, invertebrates or plants or increased invertebrate diversity. Two replicated studies (one controlled, paired) from Ireland and the UK found mixed or no differences in invertebrate numbers between created ponds and controls or natural ponds. Scrapes improve foraging conditions for breeding northern lapwing *Vanellus vanellus* (Eglinton et al. 2010).

3.10.4.2 Co-Benefits and Trade-offs

None.

3.10.4.3 Magnitude

The spatial scale of scrapes is always likely to be small, so effects will be small, but they could have significant local effects on habitat suitability for species such as lapwing.

3.10.4.4 Timescale

Scrapes could be used by birds in the same season in which they are created, but possible benefits for demography and population size will only be realised after multiple years, when a few generations of birds provide cumulative increases in numbers.

3.10.4.5 Spatial Issues

Scrapes needs to be placed where they complement other, existing components of target species' habitats and hence support bird breeding at the site.

3.10.4.6 Displacement

This will be small as scrapes will not have a large footprint and will probably be placed in damp areas where crop or grass productivity is already low.

3.10.4.7 Maintenance and Longevity

Scrapes may need to be re-established periodically, say every few years.

3.10.4.8 Climate Adaptation or Mitigation

Scrapes can play a role in making habitats suitable for species and hence facilitate managed adaptation.

3.10.4.9 Climate Factors / Constraints

None

3.10.4.10 Benefits and Trade-offs to Farmer/Land-manager

None

3.10.4.11 Uptake

N/A

3.10.4.12 Other Notes

None

3.10.5 ETPW-198 Use targeted habitat management for species with highly specialised requirements and ETPW-211 Undertake targeted measures to recover populations of rare, threatened or otherwise vulnerable species (e.g. a package of targeted measures that meet the year round life cycle requirements of a species)

These 'actions' cover broad suites of very specific interventions and refers to a general philosophy rather than a coherent process or activity. Target species are likely to be uncommon and/or range-restricted, and highly diverse in their requirements.

3.10.5.1 Causality

The management involved here could cover whole farms or landscapes, or could be conducted within field boundary vegetation or bespoke structures (as **EBHE-302** Install/ maintain bird and insect houses), so it is impossible to generalise. In all cases, however, success of such actions will depend upon the management being effectively at addressing the factor(s) that limit presence or abundance of the target species. This means that a sound evidence base for all target-action combinations is essential, or that actions are undertaken in an experimental context, in which case adequate counterfactuals with no management action need to be included, together with a monitoring programme. A review cannot be conducted here without identification of specific targets and/or actions. The extent to which evidence quality supports the implementation of suitable, effective management will vary considerably between species, because previous research effort is likely to have been very uneven. This is still more relevant for the example of targeted measures for year-round life cycle support, as it is unlikely that resources for all parts of the life cycle are equally deficient in a single location: one factor is likely to be the key limit at any given time and place, so aiming to provide resource for a complete lifecycle by additional management would be inefficient. This review can only consider the overall logic chain of good, well-targeted management based on a sound evidence base and applied at a suitable spatial scale.

3.10.5.2 Co-Benefits and Trade-offs

Not assessed.

3.10.5.3 Magnitude

Not assessed.

3.10.5.4 Timescale

Not assessed.

3.10.5.5 Spatial Issues

Not assessed.

3.10.5.6 Displacement

Not assessed.

3.10.5.7 Maintenance and Longevity

Not assessed.

3.10.5.8 Climate Adaptation or Mitigation

Not assessed.

3.10.5.9 Climate Factors / Constraints

Not assessed.

3.10.5.10 Benefits and Trade-offs to Farmer/Land-manager

Not assessed.

3.10.5.11 Uptake

Not assessed.

3.10.5.12 Other Notes

None

3.10.6 ETPW-200x Provide nesting and roosting sites (e.g. fallow plots/areas for ground nesting birds and invertebrates)

A number of diverse species of conservation concern that are associated with arable land require access to bare soil and benefit from sparse and diverse vegetation for cover and/or food resources. Birds such as skylark *Alauda arvensis*, lapwing *Vanellus vanellus* and stone-curlew *Burhinus oedicnemus* require bare ground that persists through the season for nesting, but also a degree of vegetation cover for protection and to support invertebrate food. Various invertebrates of conservation value in their own right, such as mining bees (Andrenidae) require open ground that remains undisturbed and unsprayed for nesting. Fallow areas can therefore provide important habitat for these species, reflecting what they would have used traditionally, as well as the resources that would be available in weedy spring crops.

3.10.6.1 Causality

There is good evidence for skylark plots from experimental trials (Morris et al. 2004), but this is less clear from monitoring of the population effects of actual implementation. Trials have also shown a negative interaction effect with margins, which may reflect the encouragement of predators, but this is not clear (Morris & Gilroy 2008).

Lapwing plots have not been used in practice as much as expected ecologically, but this may reflect poor implementation. Stone-curlew plots are well-supported by evidence from trials (Evans & Green 2007).
Bees and herbivore-predating wasp numbers are enhanced by fallow land (Holzschuh et al. 2010). Bee richness peaks in 2-year-old set-aside fallows as annual and perennial flowers are present, suggesting a benefit from avoiding annual cultivation (Steffan-Dewenter & Tscharntke 2001). MacDonald et al. (2012) found that stone curlew plots supported more brown hares *Lepus europaeus*, vascular plants, butterflies and bumblebees, but not more carabid beetles, and also more individuals of other farmland bird species, including lapwing. However, invertebrates are not specified as targets of this option in current agri-environment schemes.

3.10.6.2 Co-Benefits and Trade-offs

Fallow plots supporting pollinator nesting may provide an enhanced pollination ecosystem service.

3.10.6.3 Magnitude

In the right context, i.e. where existing cropping is intensive and does not provide fallow land for the relevant taxa to obtain the resources that they need, these options could be very significant in allowing larger populations to persist.

3.10.6.4 Timescale

Habitat selection of new habitat patches could occur within a season, but demographic and population responses are likely to take several years to reach cumulative, detectable effects.

3.10.6.5 Spatial Issues

Fallow areas need to be created in areas/cropping systems where equivalent habitats are rare or absent to have the greatest effect. For example, skylark plots in spring cereals or farmland where other fields are fallow or spring-sown are unlikely to affect densities significantly.

3.10.6.6 Displacement

This will be negligible as the spatial footprint of fallow areas can be small.

3.10.6.7 Maintenance and Longevity

These plots need to be created annually.

3.10.6.8 Climate Adaptation or Mitigation

A small benefit may be provided by plots improving habitat quality and hence mitigating climate change impacts.

3.10.6.9 Climate Factors / Constraints

N/A

3.10.6.10 Benefits and Trade-offs to Farmer/Land-manager

These are measures with very low impact on farm operations, so achieve a conservation outcome with minimal inconvenience.

3.10.6.11 Uptake

Some farmers do not like the untidy appearance of fallow plots within cropped fields.

3.10.6.12 Other Notes

None

3.10.7 ETPW-260x Provide feeding areas to support the lifecycles of wild birds (eg wild bird seed mix)

This action is interpreted as referring to wild bird seed mix and direct supplementary feeding, as it is not clear what other interventions could fit this description. The former is a sacrificial crop mixture sown specifically to feed birds, left to set seed and only cut or ploughed once the seed has been exhausted, typically in February, although some mixtures are designed to be biannual, with crops such as kale or fodder radish providing seed in a second winter. Other crops in these mixtures include barley, triticale,
kale, quinoa, linseed, millet, mustard and sunflower. Maize is excluded. Direct supplementary feeding consists of the provision of seed directly to birds, usually by spreading along a track or other hard, non-cropped substrate. It is particularly intended to provide food for birds during the late winter ‘hungry gap’ period.

3.10.7.1 Causality

Wild bird seed mix can benefit priority birds by providing seed-bearing crops over winter. Wild bird cover is associated with increased abundance, density and species richness of priority farmland birds, including grey partridges (Aebischer et al. 2000), skylarks, finches and buntings (Boatman et al. 2003; Henderson et al. 2004; Stoate et al. 2004; Vickery et al. 2009). Crop mixes dominated by or including kale are used most widely by the greatest range and number of farmland birds; maize is avoided and has hence not included in agri-environment seed mix prescriptions (Boatman et al. 2003; Henderson et al. 2004). Wild bird cover crops are preferred habitat of priority skylarks and yellowhammers (Boatman & Bence 2000; Murray et al. 2002). Redhead et al. (2018) showed that provision of wild bird cover was associated with increased winter abundance that also enhanced breeding abundance of priority seed-eating finches and buntings at the farm scale. Regional variation in benefits of cover crops for birds have been reported, with increased winter abundance some regions but not others (Field et al. 2010a), and provision of cover crops did not prevent the decline in abundance of grey partridges on farms in one study (Browne & Aebischer 2003). In monitoring of the effects of implementation of this option at the national scale in England, Baker et al. (2012) found net positive effects on population growth rates across a suite of seed-eating birds, although a repeat analysis with five more years of data found that the patterns were more equivocal (Pringle & Siriwardena 2018), possibly because of attraction of predators or disease to the seed mix plots (Bro et al. 2004). Dicks et al. (2015) reported that 15 studies (including a systematic review) from the UK found fields sown with wild bird cover mix had more birds or bird species than other farmland habitats. Six studies (including two replicated trials) from the UK found birds used wild bird cover more than other habitats. Nine replicated studies from France and the UK found mixed or negative effects on birds. Eight studies (including two randomised, replicated, controlled studies) from the UK found wild bird cover had more invertebrates, four (including two replicated trials) found mixed or negative effects on invertebrate numbers. Six studies (including two replicated, controlled trials) from the UK found wild bird cover mix benefited plants, two replicated studies did not. Staley et al. (2016) concluded that there was strong evidence for effects and impact of this option on birds overall, but on other taxa.

Some specific seed mixes have been developed to combine the benefits of seed for birds and nectar sources for pollinators, hence benefiting both groups (Hinsley et al. 2010, Nowakowski & Pywell 2016).

For wild bird seed mix, overall, there is clear habitat selection evidence showing that food resources are provided effectively, but more mixed evidence of population growth rate benefits: there have been generally positive effects at the national scale initially, but diminishing effects over time for unknown reasons. However, it is important to recognise that the level of evidence and extent of testing for this intervention-taxon combination is far greater than those achieved in most cases – for example, for most options and taxa, there has been no national-scale monitoring of population responses to management in practice.

This option is not designed to support bats, and McHugh et al. (2018) found no preference among AES option types including wild bird seed mix for three pipistrelle bat species.

There have been few studies of direct supplementary feeding in practice, but there is strong experimental evidence of positive effects of selection in the field and on population growth rates (Siriwardena et al. 2006, 2007), especially when food is provided in late winter (Siriwardena et al. 2008). It is possible that feeding could lead to negative effects from the facilitation of disease transmission.

3.10.7.2 Co-Benefits and Trade-offs

Not assessed

3.10.7.3 Magnitude

The effects can be large and could contribute significantly to population growth.

3.10.7.4 Timescale

Behavioural, habitat selection effects are apparent in the year of sowing, but demographic effects are likely to be strongest in harsh winters or when food is scarce, so multiple years may be needed for effects on survival to be manifested.

3.10.7.5 Spatial Issues

Seed provision will have the greatest effect in landscapes where ambient winter food resources are most scarce (for example, low densities of weedy stubble fields), although this may be difficult or impossible to predict from available data sources.

3.10.7.6 Displacement

Crop patches cover small areas in fields, so displacement of production is possible but probably not large in extent.

3.10.7.7 Maintenance and Longevity

Crops need to be sown annually or biannually, but indefinitely.

3.10.7.8 Climate Adaptation or Mitigation

N/A

3.10.7.9 Climate Factors / Constraints

Crop mixes may need to be revised to ensure good seed yields after changes in climate.

3.10.7.10 Benefits and Trade-offs to Farmer/Land-manager

Many farmers use seed crops to support gamebird shoots, which may attract them as they may be paid for the land used to do this.

3.10.7.11 Uptake

Direct supplementary feeding brings with it some risk of crop volunteer ‘weeds’ and both this and wild bird seed mix crops can attract rat populations, which may put farmers off.

3.10.7.12 Other Notes

None

3.10.8 ETPW-260y Provide feeding areas to support the lifecycles of pollinators (eg pollinator seed mix)

Pollinator seed mixes or nectar flower crops are sacrificial cropped areas sown specifically to provide pollen and nectar resources for insects – notably bees, hoverflies, butterflies and moths. Mixtures feature multiple nectar-rich plants (e.g. red clover, alsike clover, bird’s-foot-trefoil, sainfoin, musk mallow, common knapweed), with no single species making up more than 50 per cent of the mix by weight. Mix plots typically need to be ploughed and re-drilled every few years to promote floral diversity.

3.10.8.1 Causality

Dicks et al. (2015) found that 41 studies (including one randomised, replicated, controlled trial) from eight countries found flower strips increased invertebrate numbers or diversity. Ten studies (two replicated, controlled) found invertebrates visited flower strips. Fifteen studies (two randomised, replicated, controlled) found mixed or negative effects on invertebrates. Seventeen studies (one randomised, replicated, controlled) from seven countries found more plants or plant species on flower strips, four did not. Overall, therefore, there were positive effects of nectar flower mixes. Similarly, Staley et al. (2016), reviewing the evidence for effects and the size of the impacts involved, concluded that there was strong evidence for a high, positive impact on pollinators, and strong evidence of a moderate impact on butterflies and birds.

Note that, while there is little population growth rate/long-term impact evidence of the kind that is available for **ETPW-260x** for the target taxa here, Image et al. (2022) report that, while ground-nesting bee populations have increased nationally, tree-nesting bumblebees, and cavity-nesting solitary bees have not. Options such as this in current schemes produce no significant increase in the national crop pollination service and only localised pollination service increases to late-spring flowering crops. This may be because there is insufficient provision of nesting resources in nectar flower mixes or complementary options.

3.10.8.2 Co-Benefits and Trade-offs

Promoting pollinators could have a positive influence on crop productivity.

3.10.8.3 Magnitude

Effects will be large where nectar resources limit pollinators, but a lack of suitable nesting habitats may limit the responses of some groups, so a couple provision of these resources is likely to be needed in many cases.

3.10.8.4 Timescale

Habitat selection effects are apparent in the year of sowing, but multiple years may be needed for population effects to be manifested.

3.10.8.5 Spatial Issues

Effects will be greatest where nectar resources are scarce.

3.10.8.6 Displacement

Crop patches cover small areas in fields, so displacement of production is possible but probably not large in extent.

3.10.8.7 Maintenance and Longevity

Mix plots typically need to be ploughed and re-drilled every few years to promote floral diversity.

3.10.8.8 Climate Adaptation or Mitigation

N/A

3.10.8.9 Climate Factors / Constraints

Crop mixes may need to be revised to ensure good seed yields after changes in climate.

3.10.8.10 Benefits and Trade-offs to Farmer/Land-manager

Promoting pollinators could have a positive influence on crop productivity.

3.10.8.11 Uptake

N/A

3.10.8.12 Other Notes

None

4 Key Action & Evidence Gaps

4.1 Actions with Assessment of Trade-off/Co-benefits only

4.1.1 EBHE-081 Minimise cultivation on Scheduled Monuments/ heritage assets on the shine database that are not Listed Buildings or Scheduled Monuments

4.1.1.0 Co-Benefits and Trade-offs

Minimising cultivation promotes soil biodiversity and many species that depend on soil invertebrates, but this specific intervention has not been researched and will only cover small areas in any given site, so will have only small effects at best.

4.1.2 EBHE-187 Create a landscape appraisal of the holding in the context of the local area to identify key characteristics that will inform integrated implementation of actions to conserve and enhance the landscape character

4.1.2.0 Co-Benefits and Trade-offs

This is a general plan for management that will lead to positive or neutral effects on biodiversity if conducted appropriately.

4.1.3 EBHE-233 Control scrub or trees to maintain views

4.1.3.0 Co-Benefits and Trade-offs

There is no specific evidence for this action, but its effects are likely to be mixed. There are two possible contexts: trees and scrub at viewpoints, where control impacts on overall vegetation in the area would be minimal, and effects on viewsheds, where entire landscapes could be changed, such as to recreate a traditionally open vista. The latter would cause negative effects on local woodland species and positive ones on local open habitat species, especially those that are negatively influenced by vertical structure in such landscapes. Nevertheless, effects are likely to be localised.

4.1.4 EBHE-316 Control scrub or trees on top or in front of geodiversity features

4.1.4.0 Co-Benefits and Trade-offs

Geodiversity features may be damaged by tree roots; presumably this refers mostly to rock and cliff sites, where potential biodiversity value is low, but certain plant species might be affected negatively. However, more species are likely to benefit from trees in these environments than to be affected negatively by them, so this action is likely to have mixed, mostly positive effects, although all with small impacts, because the areas involved will be small.

[TOCB Report-3-6 Carbon **EBHE-316**] There is good evidence that the long-term reduction of vegetation biomass will reduce above ground carbon stocks, and result in a carbon debt, despite a potentially increase rates of sequestration by remaining vegetation (Gregg et al., 2021; Matthews et al., in prep.). Management to remove vegetation can also cause soil disturbance and the loss of below ground carbon stocks depending on the level of control (Matthews et al., in prep.). Allowing an increase in total vegetation biomass long term will increase total terrestrial carbon stocks. However, the removal of invasive species could facilitate regeneration by native species with greater carbon storage potential. For more detail see carbon review section 3.11.10 and 3.14.3.

Global, regional & local climate regulation	Above ground carbon sequestration	**
Global, regional & local climate regulation	Below ground carbon sequestration	*

4.1.5 ECCA-016 Create/ enhance/ manage Sustainable Drainage Systems on farmers own land

4.1.5.0 Co-Benefits and Trade-offs

SuDS have mostly been conceived and studied in an urban context, but can also apply within farmland, as are a collection of physical structures used to mimic natural processes (Avery 2012). Rural SuDS slow down or prevent the transport of pollutants to watercourses by breaking the delivery pathway between the pollutant source and the receptor. They also temporarily capture water and slow down flow, providing valuable aquatic habitats in the form of micro-wetlands for farmland wildlife. There is little direct evidence of biodiversity benefits (Shaw et al. 2015), but increasing habitat heterogeneity by providing new, wet or damp habitat resources will benefit various plant, invertebrate and bird species (subject to implementation at sufficient scale), assuming that the habitats that they replace are more intensively managed farmland (Bradbury & Kirby 2006).

4.1.6 ECCM-075 Install agrivoltaics

4.1.6.0 Co-Benefits and Trade-offs

Agrivoltaic landscape designs can provide habitat for groups such as pollinators and improve biodiversity (Toledo & Scognamiglio 2021). In dry, hot, water-restricted systems, shading from agrivoltaics can increase biodiversity in the form of flowers and polinators (Graham et al. 2021), and the potential for vegetation around agrivoltaic panels to support biodiversity also exists in Europe, although it is not yet well-studied and is likely to be affected by how the habitat is managed (van Aken et al. 2021, Trommsdorff et al. 2021). Replacing agricultural land-use, the physical structures of agrivoltaics are likely to reduce suitability for birds and mammals of open habitats, but the extent of the effect will depend on the nature of the habitat that is replaced.

4.1.7 ECPW-046 Create, enhance or manage channels

4.1.7.0 Co-Benefits and Trade-offs

This is too vague to assess meaningfully for biodiversity: the purpose is unclear, so the meaning of ‘enhancement’ is unknown, and the habitat context of the ‘channels’ is not specified – assuming that this concerns drainage systems, the evidence considered under **ECCA-016** should be relevant. Alternatively, it could refer to floodplain management (**EBHE-126**).

4.1.8 ECPW-048 Create, enhance or manage water retention ponds

4.1.8.0 Co-Benefits and Trade-offs

These will have limited benefit for biodiversity, as they are assumed to be managed specifically for water retention.

4.1.9 ECPW-059 Reconnect rivers with floodplains

4.1.9.0 Co-Benefits and Trade-offs

Allowing rivers to flood natural floodplains in periods of high flow would influence land-use by making intensive farming of that land less feasible and by producing wetter, grassland habitat that potentially provides habitat for wading birds and plants that are associated with damp soil, as well as the invertebrate assemblage that is found with this vegetation. Significant negative effects are unlikely. However, the size of the benefits depends on factors such as flooding frequency, flood timing and duration, water depth, water quality and temperature, and management (grazing or mowing) (Ibe et al. 2014). Floodplain meadows can be recreated in terms of vegetation composition, but this can take 70-150 years (Woodcock et al. 2011).

[TOCB Report-3-6 Carbon **ECPW-059**] When considering the hydrological restoration of the floodplain, there is good evidence from sites across Europe that restoration will result in significant local carbon sequestration in sediments (Gregg et al., 2021). Approximately 42% of floodplains in England and Wales are no longer connected to river systems (Gregg et al., 2021), and high rates of intensive agricultural land use occur on floodplains in the UK which has a strong negative impact on carbon storage in floodplain soils, and have no long-term above ground carbon storage (Gregg et al., 2021).

Following hydrological restoration, initial rates or carbon sequestration are particularly high, with sequestration rates of -1 tC ha-1 yr-1 estimated for the first century following restoration along parts of the Danube in Austria (Zehetner et al., 2009). Although rates sequestration decrease exponentially, carbon stocks in floodplains are relatively high compared to other terrestrial habitats, with 109.4 t C ha-1 found in the top 10 cm of soil in Cricklade National Nature Reserve (Gregg et al., 2021). More widely, estimated values for temperate floodplain carbon storage are on the orders of magnitude of 100-1000 tC ha-1 with reported accumulation rates of -0.1 to -3 tC ha-1 yr-1, which are affected by climate, geology, land cover, and fluvial properties (Sutfin et al., 2016). A significant proportion of carbon sequestration in floodplains is allochthonous POC that is deposited when floodplains are inundated (Cook, 2007). This means that the increase or decrease of below ground carbon in floodplains is not a reliable indicator of carbon gain or loss at the landscape scale, as reduced deposition may be due to improved erosion protection upstream, and increased erosion does not necessarily mean carbon has not been sequestered downstream, for example in saltmarsh. As such, the additionally of floodplain carbon stores is associated with a large amount of uncertainty (Quinton et al., 2010). Lastly, significant methane emissions have be documented in wetlands due to microbial respiration in anoxic conditions, which may offset some of the benefits form wetland restoration and reduced agricultural emissions (IPCC Task Force on National Greenhouse Gas Inventories, 2014).

Food and fibre production	Area under production or yield and outside of ELM	**T***
Global, regional & local climate regulation	Above ground carbon sequestration	**T***
	Below ground carbon sequestration	**LT****

4.1.10 ECPW-060 Create, enhance or maintain river habitats

4.1.10.0 Co-Benefits and Trade-offs

Biodiversity benefits of river enhancement primarily apply within the river channel, but there are some benefits to terrestrial biodiversity, notably from aquatic or part-aquatic invertebrates providing food resources (e.g. Paetzold et al. 2005, Collier et al. 2002), and reducing pollution, for example, could have positive effects on terrestrial insectivores (Kraus 2019, Kotalik 2020).

4.1.11 ECPW-276 Use biostimulants from recommended list to reduce the need for artificial N

4.1.11.0 Co-Benefits and Trade-offs

This is a general nutrient management action, from which we expect an overall reduction in nutrient inputs, with various positive effects on biodiversity. However, effects will depend on the context of baseline nutrient levels.

4.1.12 ECPW-277 Reduce and reuse plastics in agriculture, forestry and horticulture

4.1.12.0 Co-Benefits and Trade-offs

This action is too general to review meaningfully for biodiversity effects. There is no evidence relating to plastic covering of soil. Effects of this practice, or stopping it, also depend critically on what the alternatives are. Most elements of plastic use are not relevant to biodiversity. The possible effects of plastic mulches are considered under **ECPW-280**.

4.1.13 EHAZ-061 Create/ enhance/ maintain earth bunds to intercept surface flow

4.1.13.0 Co-Benefits and Trade-offs

This measure would create wet habitats on farmland and hence resources including: (i) damp soil, for probing bird species; (ii) permanent water to provide water-dependent invertebrates, as a source of food; (iii) bare or sparsely vegetated ground in the draw-down zone, to improve access to food; (iv) rank emergent vegetation for nesting birds (Bradbury & Kirby 2006). This has not been tested, however, beyond a single project (Bailey et al. 2007).

4.1.14 EHAZ-102 Harvest and store rain water to reduce reliance on abstraction

4.1.14.0 Co-Benefits and Trade-offs

This is a general, long-term and large-scale proposed change in practice; the evidence for biodiversity effects can only hypothetical/logic-based. Any benefits will mainly be for species in watercourses, not terrestrial biodiversity.

4.1.15 EHAZ-108 Store surface water over permeable ground (for aquifer recharge)

4.1.15.0 Co-Benefits and Trade-offs

M **EHAZ-061** – It is likely that the same effects on soil and terrestrial species' responses will be relevant.

4.1.16 EHAZ-110 Use trickle or drip irrigation

4.1.16.0 Co-Benefits and Trade-offs

Not reviewed –more efficient use of water for irrigation but no effect on soil/biodiversity. Benefits would be related to abstraction and limited to watercourses.

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