Qualitative Impact Assessment of Land Management Interventions on Ecosystem Services ("QEIA")

Report-3 Theme-5C:
Biodiversity - Semi-Natural Habitats

30-June-2023

Qualitative impact assessment of land management interventions on Ecosystem Services

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Citation

How to cite

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)

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)

Newell Price, J.P., Williams, A.P., Bentley L. & Williams, J.R. (2023). *Qualitative impact assessment of land management interventions on Ecosystem Services (“QEIA”)*. Report-3 Theme-3: Soils (Defra ECM_62324/UKCEH 08044)

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

This section covers semi-natural habitats and largely excludes those managed for agricultural production (cropland and grassland) but does include upland areas which may be grazed. Woodlands are considered here along with coastal habitats, uplands, heathlands, wetlands and freshwater habitats.

Review methodology: Much of the evidence in this section has been amassed from existing reviews and meta-analyses of options, either in the scientific literature, in reports (e.g., reporting on options for ERAMMP Environment and Rural Affairs Monitoring and Modelling Programme www.erammp.wales) or, for example, using the Conservation Evidence website www.conservationevidence.com. For actions with little published peer-reviewed literature, we used expert opinion and, where available, existing guidance from NGO's. The evidence reviewed was used to attribute scores on the likely relative benefit / dis-benefit for each action, in relation to the studied measures of biodiversity, as compared to cases with no action. Where information was readily available and clearly appropriate, we have added evidence to the various headings (e.g., magnitude, spatial issues, displacement, etc.). However, in some cases this information was hard to disentangle from the existing evidence, not available, or we did not have adequate resource to investigate evidence under each of these headings. In all cases we have scored actions on the assumption that they have been 'done well'. Clearly, inadequate implementation of actions could result in differing outcomes.

2 Outcomes

Table 1: The individual 'indicators' against which we were asked to score each action in relation to their impacts on biodiversity, non-native species, pest and disease control and pollination and seed dispersal. Small 'feature' habitats have been interpreted as small areas of semi-natural habitat isolated wider agricultural landscapes (may include, e.g. hedges, ponds, clumps of trees). 'National species occurrence' refers to increasing the occurrence of INNS at a national scale. Priority species were defined as those with Section 41 status in England (2006 Natural Environment and Rural Communities Act), or for those taxa with more recent red lists, as vulnerable, threatened, endangered or critically endangered. Cells highlighted in grey emphasise the indicators for which actions were scored for the ecosystem services (ES) relating to wider biodiversity (i.e. not priority species or protected areas), as this was considered the most relevant to agri-environment actions.

"Service"	Indicators for ecosystem services flow
Biodiversity	Biodiversity adaptation - maintaining / enhancing biodiversity under a changing climate
	Atmospheric deposition of N and exceedance of critical loads
	Connectivity of small 'feature' habitats
	Enhance condition of agricultural land (*interpreted as 'enhance abundance and/or species richness of **wider** farmland biodiversity'*)
	Enhance condition of semi-natural habitat
	Favourable condition of SSSIs
	Maintain good condition agricultural land (*interpreted as 'maintain abundance and/or species richness of **wider** farmland biodiversity'*)
	Maintain good condition of semi-natural habitat
	Presence of rare (red list) species; Presence of priority species
INNS Invasive Non-Native Species [1]	National species occurrence

[1] https://uk-scape.ceh.ac.uk/our-science/projects/GBNNSIP

Pest and disease control	Evidence of outbreaks of pests and disease
Pollination and seed dispersal	Increased abundance, distribution & species richness of pollinators & seed dispersers

Table 2: The Integrated Assessment scoring and coding system

Cell Code		
R	Already covered by regulation; not assessed further	
B or S	Too big to review so split and reviewed under individual actions	
M	Merged actions can be reviewed together for a specific outcome as evidence does not support more granular approach	
N or no letter	Not relevant / has no impact on ES	
X	Topic not included by current review teams	
Colour Code	**Magnitude score (*)**	**Contextual issues (Letters)**
Green		
Well tested at multiple sites with outcomes consistent with accepted evidence logic chain. No reasonable dis-benefits or practical limitations relating to successful implementation. Plus:	*** Action can have major benefit if done well ** Action can have moderate benefit if done well * Action has limited benefit even if done well	'T' only used for two types of action: • Creation of plans • Monitoring and measurement
Amber		
Evidence is currently limited for the impact on this ES indicator and / or there may be some disbenefits embedded within this indicator (e.g. different taxa response differently) and /or it is contextually dependent for each specific ES indicator	*** Action can have major positive benefits on this ES indicator if done well ** Action can have moderate positive benefit on this ES indicator if done well * Action has limited impact on this ES benefit even if done well	'L' Limited evidence for benefit but is consistent with evidence logic chain 'T' Contextually dependent benefits and / or requires targeting to be effective 'D' There are some disbenefits to some services within this ES indicator. *Note*: no impact does not make a box an amber if all other sub-indicators for this ES indicator are green – it should then be a green if nothing disbenefits.
Red		
A disbenefit known for this action for this ES indicator and / or limited evidence but evidence logic chain suggests a highly likely disbenefit.	*** Action can have major disbenefit on this ES indicator ** Action can have moderate disbenefit on this ES indicator * Action has limited impact disbenefit on this indicator	'T' Only used where there was a context dependency for the Food and Fibre ES.
Grey – duplicate action		

Evidence was limited for the outcome *Biodiversity adaptation - maintaining / enhancing biodiversity under a changing climate*. While agri-environment scheme (AES) actions could theoretically support biodiversity adaptation under climate change, there is little empirical evidence of this, except in the specific case of the creation of coastal habitats as a result of climate change impacts on coasts. If there was evidence that the action enhanced or maintained abundance or species richness of wider biodiversity, this may lead to more resilient communities of farmland biodiversity. The 'b*iodiversity adaptation'* outcome was therefore scored according to the impacts on wider biodiversity, but with a maximum score of Amber L (with stars) due to limited evidence. *Atmospheric deposition of N and exceedance of critical loads* is relevant to biodiversity of semi-natural habitats since increases of N in these habitats can cause a shift in biodiversity towards more nutrient loving species, thereby displacing species typical of the semi-natural habitats. Where relevant (particularly in upland habitats) this has been scored for semi-natural habitats.

3 MANAGEMENT BUNDLES

Options have been aggregated under the following management bundles (all options are listed under each sub-heading)

- **Habitat Creation (3.1-3.7)**
  This management bundle concerns the creation of semi-natural habitats, at least those which can be ‘created’ by land managers. All habitats take time to mature and whilst habitats are constantly evolving there are recognised states which enable classification into habitat types. Some habitats, such as peat bogs, take centuries to develop. This bundle covers management actions which initiate the creation of a range of habitat types.

- **Actions for habitats with specific hydrological characteristics (3.8-3.11)**
  This management bundle includes actions for habitats which would naturally be shaped by their underlying hydrology, in particular bogs, fens and flood meadows.
- **Restoration management and enhancement of semi-natural habitats (3.12-3.14)**
  This management bundle is focused on the restoration, management and enhancement of semi-natural habitats. It incorporates: Coastal, Grassland, Mountain, Moor and Heath, Riparian, Rivers/water courses, Woodland and Woody features.

- **Natural regeneration (3.15-3.17)**

- **Maintenance and restoration of habitat features in parks and gardens (3.19)**
  The Historic England 'Register of Parks and Gardens of Special Historic Interest in England', established in 1983, currently identifies over 1,600 sites assessed to be of particular significance.

- **Systems actions/mixed systems and cross-habitat actions (3.20)**

- **Specific wildlife targeted actions (3.21)**

N.B. All actions for ponds (including management and enhancement) are covered under Habitat creation – Ponds/water bodies and wetlands (section 3.4).

3.1 HABITAT CREATION - COASTAL

3.1.1 ETPW-081C, ECCA-033C, ECCM-043C, ETPW-180C, ETPW-081CX, EHAZ-070C & ETPW-179C

ETPW-081C	Create coastal habitats (split)
ECCA-033C	Create coastal habitats to compensate for losses to climate change as part of a coastal management plan (assessed)
ECCM-043C	Create coastal wetland habitats
ETPW-180C	Create inter-tidal and saline habitats
ETPW-081CX	Create salt marsh
EHAZ-070C	Create sand dunes
ETPW-179C	Create shingle features

3.1.1.1 Causality

ETPW-081C Create coastal habitats has been split because most of the evidence for impacts on biodiversity are for individual habitat types. **ECCA-033C** to create coastal habitats to compensate for losses to climate change and using a coastal management plan has been assessed (**amber**) with little actual evidence, but likely to be very positive (*****). The creation of individual habitat types in coastal systems is not possible without consideration of the complex system. A holistic approach is required (Jones et al. 2021) to understand how the dynamism of the system including hydrology, geomorphology, non-coastal land uses and climate, interact with management to create coastal habitats. It would be inappropriate to create localised habitat without considering these system interactions. Coastal Habitat Management Plans (CHaMPs) should be used to assess which sites are at risk from saline intrusion and whether habitat creation or assisted migration is required (Natural England & RSPB 2019).

ETPW-180C and ETPW-081CX - There is considerable evidence **(green**) for positive impacts of the creation of inter-tidal and saline habitats through managed realignment on coastal erosion and vulnerability of coastlines to climate change. Managed realignment refers to the breaching of existing coastal embankments originally constructed to allow saltmarsh to be converted to agricultural land. Much of the restoration/creation of inter-tidal and saline habitats has been on former intertidal areas drained for agriculture. With relatively little management, intertidal mudflats will develop and if the elevation is suitable salt marsh plants will colonise (Garbutt et al. 2006; Garbutt and Wolters 2008) **providing a valuable habitat for biodiversity**. However, comparisons of restored and existing marshes (reference communities) have shown considerable differences, (Garbutt et al. 2006; Garbutt and Wolters 2008). Some species, such as sea lavender (*Limonium vulgare*), sea plantain (*Plantago maritima*) and sea pink/sea thrift (*Armeria maritima*) tend to be rare on restored marshes (Hudson et al. 2021). In a survey of de-embankment of historically reclaimed salt marshes many sites contain less than 50% of the regional target species, especially when sites are smaller than 30 ha. Higher species diversity is observed for sites exceeding 100 ha and for sites with the largest elevational range within mean high-water neap to mean high-water spring tide (Wolters et al. 2005). Although less commonly recorded, nationally scare plants such as golden samphire *Inula crithmoides*, shrubby seablight *Suaeda vera* and small cord-grass *Spartina maritima* have all been recorded within realignment sites (Garbutt 2005). Where salt marsh develops with a sparse cover of vegetation, large numbers of golden plover (*Pluvialis apricaria*) and lapwing (*Vanellus vanellus*) typically use these areas for roost sites (Garbutt & Boorman 2009, Mander et al., 2007). More information on the colonisation of bird and invertebrate species, for which these habitats are particularly important, is provided below (Timescale).

Relatively rapid colonisation may be expected from pioneer and low-marsh species, provided they are present in a nearby source area and the restoration site is at the appropriate altitude (Wolters et al. 2005, Erfanzadeh et al. 2010). To enhance diversity further, additional work may be required through introducing source material to the sites. Planting might be appropriate to help establish vegetation in estuary fringing habitat where conditions prevent seedling establishment (Hudson et al. 2021). Restoration success may also be limited by the amount of land available to enable the full zonational range of saltmarsh communities (Wolters et al. 2008).

Where sites are at present too low for the development of saltmarsh, mudflats have been colonised by intertidal invertebrates providing additional feeding areas for wading birds (Atkinson, 2004). Fish have also been recorded using realignment sites for feeding and refuge (Garbutt & Wolters 2008). Managed realignment tends to create high-level mud in the early phases of development, and observations from several sites suggest that some species of wading bird such as dunlin (*Calidris alpina*) and redshank (*Tringatotanus*) use these areas to provide additional feeding time, either side of high tide.

EHAZ-070C has been re-merged with **EHAZ-070EM** to create **EHAZ-070.** For the UK it was not possible to find any evidence for the creation of sand dunes through an agri-environment option and indeed creation of sand dunes is not an option for which much evidence is yet available internationally. A project called Sand Motor 2 in the Netherlands which fed huge volumes of sand extracted from the seabed (at a cost of 70M euros) into an existing sand dune system in 2011 to improve coastal protection will produce a 10 year report this year. Early results indicated that the new sand shoal created through the Sand Motor, offers new habitat for flora and fauna, especially in and around the sheltered and shallow part of the lagoon.

ETPW-179C - The evidence for creation of shingle is **amber/red**. There does not appear to be any recent evidence beyond a comprehensive Natural England report (Doody & Ranwell 2003). Beach nourishment with shingle is frequently used to improve sea defence capability. However, although it can form a foundation for vegetation restoration it frequently conflicts with habitat conservation in the short-term particularly in areas where dumps of gravel on existing stable beach forms can destroy both ephemeral strandline plant communities-with associated and sometimes rare species of both plants and invertebrates and more stable structures and their communities.

Creating/restoring the physical shingle structure (ridges and lows) is difficult, especially on dry well drained shingle and there is little evidence of effective artificial recreation of the surface shingle form of ridges and hollows. Doody & Ranwell (2003) suggest that regrading shingle is feasible but takes time. The National Trust undertook to test, experimentally, whether it was possible to regenerate shingle flora on some of the worst degraded and damaged sites. The project was carried out in 2000 as part of the European Union LIFE-Nature project 'WILD NESS - The Conservation of Orfordness, Phase 2'. The reestablishment of vegetated shingle does not seem to readily occur even with the introduction of seed. The natural forces of tides and waves are much more efficient tools for sorting coastal shingle than anything human restoration can achieve.

3.1.1.2 Co-Benefits and Trade-offs

[TOCB from Report-3-6 Carbon **ECCA-033C**] Coastal habitats are able to sequester and store significant amounts of carbon below ground, but the capacity for this varies substantially across the coastal habitat types in the UK. For reviews of the potential of coastal habitat creation to result in carbon sequestration see the QEIA Carbon Sequestration Report-3-6. Climate change is a significant risk to coastal habitats and their carbon store and sequestration potential, as a result of sea level rise, changing temperatures and storm severity and frequency (Macreadie et al., 2019). These losses could be offset by the creation or restoration of coastal habitats elsewhere through managed realignment (Boorman & Hazelden, 2017), or by allowing the landward expansion of existing coastal habitats as sea levels rise. The evidence base for these activities, their feasibility and their impact on carbon sequestration are highly variable across habitats. The creation and restoration of highly degraded salt marsh has good supporting evidence for a positive impact on carbon sequestration potential (Gregg et al., 2021; Parker et al., 2021). Evidence for the net impact of the landwards migration of coastal habitats on carbon sequestration is limited and based on expert opinion.

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**

[2] https://climate-adapt.eea.europa.eu/metadata/case-studies/sand-motor-2013-building-with-nature-solution-to-improve-coastal-protection-along-delfland-coast-the-netherlands

3.1.1.3 Magnitude

ETPW-180C - By 2018, 50 managed realignment schemes had been completed in the UK creating almost 2500ha of habitat (Hudson et al. 2021) for biodiversity, preventing coastal erosion and reducing the vulnerability of coastlines to climate change.

3.1.1.4 Timescale

ETPW-180C - Vegetation cover is usually established within 5 years - but differences in species composition and abundance may be considerable (Garbutt et al. 2008, Wolters et al. 2005). Surveys of regenerated saltmarsh on sites that were breached over 100 years ago as the result of storm events have shown that even after 100 years species composition differs from adjacent 'natural' saltmarsh (Wolters et al. 2008, Garbutt and Wolters 2008).

The time taken for invertebrates to colonise the sediment at a site will be affected by their life history and availability in the local species pool. Species that are mobile as adults are recorded in the early stages of colonisation. For example, the crustacean European mud scud *Corophium volutator* regularly occurs in large numbers at several sites (Atkinson et al., 2001). More sedentary species, such as bivalves, rely on planktonic stages to colonise new sites and it may be several years before a stable population of these relatively long-lived species becomes established (Garbutt & Boorman, 2009). These invertebrates are key prey items for wading birds including avocets (**Recurvirostra avosetta)** and godwits. Above ground invertebrates, including spiders, have been shown to colonise rapidly, quickly restoring species richness, although equivalent composition may take decades (if at all) because of differences in the composition of the newly generating vegetation (Petillon & Garbutt 2008).

Birds are quick to colonise, either for roosting or to feed. In the first few years after realignment, waterbird assemblages are generally variable and undergo large changes adjusting to the biological and physical evolution of the site (Atkinson et al., 2004). The first few years are characterised by a dominance of passerine species such as sky lark (*Alauda arvensis*), meadow pipit (*Anthus pratensis*), and reed bunting (*Emberiza schoeniclus*). Mander et al. (2007) found that within 3 years of creation, the Paull Holme site (UK) was capable of supporting a functional waterbird assemblage of similar composition to that of adjacent existing intertidal areas at low water (Garbutt & Boorman 2009).

3.1.1.5 Spatial Issues

ETPW-180C - Much of the restoration/creation of inter-tidal and saline habitats has been on former low-lying intertidal areas drained for agriculture.

EHAZ-070 - Only relevant in areas with existing coastal sand/shingle deposition.

3.1.1.6 Displacement

ETPW-180C - Managed realignment leads to some loss of agricultural habitat and production capacity.

3.1.1.7 Maintenance and Longevity

ETPW-180C - This action is a permanent land use change away from agriculture. However, sediment movements are dynamic and salt marsh areas can continue to decrease in area as a result of erosion.

EHAZ-070C and **ETPW-179C** - Sediment movements are dynamic and dependent on physical changes which may not be manageable.

3.1.1.8 Climate Adaptation or Mitigation

ETPW-180C, EHAZ-070C and ETPW-179C - Sea level rise is a threat to coastal habitats. Creation of more habitat is beneficial to coastal protection, ecosystem function, carbon storage and biodiversity.

3.1.1.9 Climate Factors / Constraints

ETPW-180C - Sites are dependent on sea levels and potentially on flow of water and sediments from estuaries which may in turn be influenced by climate factors. The Natural Environment chapter of the **UK Climate Change Risk Assessment Evidence Report** (Brown, et al 2016) highlights that all coastal ecosystems are at high risk from climate change, due to the presence of flood defence and erosion protection structures, which prevent landwards rollback of the intertidal zone as a natural response to sea-level rise. Natural adaptive capacity is also limited by reduced sediment supply due to hard coastal defences (NE and RSPB, 2019).

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

ETPW-180C - Loss of agricultural land may be a trade-off though low intensity grazing on salt marsh may improve plant diversity and provide breeding habitat for birds (see section 3.12.2). There could also be recreational and tourism benefits. In addition, salt marsh products can sometimes attract a premium. Salt marsh habitats may also be used for gathering of samphire which can be harvested, sold and eaten as a delicacy (Hudson, Kenworthy and Best 2021).

There may be recreation and tourism benefits.

3.1.1.11 Uptake

ETPW-180C - In areas where breaches of sea defences threaten agricultural land farmers may be left with little choice but to accept that their land is no longer going to be suitable for management in the same way. Successful schemes have worked closely with landowners to manage their needs.

3.1.1.12 Other Notes

N/A.

3.2 Habitat Creation – Mountain, moor and heathland

3.2.1 ECPW-176C: Create heathland (including heathland mosaics)

There is currently insufficient evidence to assess this action in terms of magnitude, spatial issues, displacement, benefits and trade-offs to farmer/land manager or uptake.

3.2.1.1 Causality

The evidence is strong/substantial (**green**) that heathland can be created/re-created. There are many studies of attempts at heathland creation/re-creation some on agricultural soils, some on acid grassland and some monitored over long time periods (Pywell et al. 2011). Heathland creation, as opposed to the enhancement or management of heathland, tends to be focused in the lowlands, hence most of the evidence presented here is for lowland heath. However, results are complex and dependent upon initial habitat and environmental conditions and context. Issues that need to be addressed, particularly if on agricultural land, include increased nutrient availability, high pH, and lack of seed sources in agriculturally improved landscapes. Addressing these issues could involve physical removal or deep ploughing of improved topsoil (Diaz et al. 2008, Allison and Ausden 2004), chemical amendment of the topsoil (Tibbett and Diaz 2005, Diaz et al. 2008, Owen and Marrs 2000) or translocation of heathland vegetation and application of seed-bearing vegetation (Pywell et al. 1996). Studies using these methods report diverse levels of success in producing long-lasting heathland and acid grassland communities.

In a comparison of a restored heathland site (through the removal of pine) with ancient sites, plant and pollinator communities were shown to have established successfully on the restored sites. There was little evidence of movement of pollinators from ancient sites onto adjacent restored sites, although pollinator species richness was similar. Pollinator networks/webs were less complex on restored sites, although a few widespread pollinators were the main pollinators on both sites indicating that functionality may be restored even if species composition is lacking (Forup et al. 2007).

3.2.1.2 Co-Benefits and Trade-offs

Not assessed.

3.2.1.3 Timescale

Pywell et al. (2011) compared different techniques for heathland restoration, in a long-term experiment, after 17 years the soil and vegetation were still not characteristic of adjacent heathland vegetation. The different treatments showed different trajectories of vegetation change. Natural colonisation by heathland species was slow due to seed limitation, resulting in the formation of an acid grassland community. After 11 years the key pollinators were the same between ancient and restored sites suggesting restoration of function although the complexity of pollinator interaction networks was reduced in restored sites (Forup et al. 2007).

3.2.1.4 Maintenance and Longevity

Maintenance of heathland requires continuous interventions (Pywell et al. 2011).

3.2.1.5 Climate Adaptation or Mitigation

Fragmentation of heathland sites will increase sensitivity to climatic effects so re-creation of new habitat particularly if connected should enhance adaptation (Natural England and RSPB 2019).

3.2.1.6 Climate Factors / Constraints

Climate change alone moderately affects plant diversity, community structure and ecosystem functions. Combined with other factors, climatic changes will condition heath development, such as seed set and seedling establishment, rare species occurrence and nutrient cycling in the soil (Fagundez 2013). Potential climate changes such as a longer growing season will affect species composition, there is likely to be an increased risk of wildfires, drought and drying out of wet heath (NE and RSPB 2019).

3.3 Habitat Creation - Grassland

3.3.1 ECPW-022C & EBHE-214

ECPW-022C Create species-rich grassland habitats
EBHE-214C Create locally distinctive flower rich meadows using traditional techniques

3.3.1.1 Causality

It is presumed that this option refers to the creation of species rich grasslands from improved/semi-improved grassland so there is considerable overlap with options **EBHE-214EM** Enhance and manage locally distinctive flower rich/hay meadows using traditional techniques and **ECPW-022EM** Enhance or manage species-rich grassland habitats.

There is good evidence **(green)** that successful diversification of grassland swards will increase plant species diversity and be beneficial for other above-ground and below-ground species including nectar plants and pollinators (Newell-Price et al. 2019, Scherber et al. 2010, Woodcock et al. 2013, Keenleyside et al. 2019). It could also be beneficial to connectivity and dispersal to have sources of species rich habitat. Sward diversity management options include increasing plant species diversity through the addition of grass, forb and legume species. This is normally carried out through field operations such as reseeding, oversowing, or slot seeding, but may also include introduction of plug plants or feeding animals with high quality hay containing seeds (from nearby sites) (Keenleyside et al. 2019, Maskell et al. 2019). Spreading green hay from nearby semi-natural sites is another method that can aid restoration of semi-natural plant communities and, by association, phytophagous insects (Woodcock et al. 2010). It may also be necessary to reduce soil fertility that can be done by soil stripping or appropriate grazing or cutting management (Bullock et al. 2011).

Increases in the richness of associated soil and foliage dwelling species such as moths, spiders and beetles are likely to be associated with these actions, but there could be decreases in the abundance of species directly associated with the previously dominant plant species. Alison et al. (2017) demonstrated that created grasslands with a higher diversity of chalk grassland wildflowers, including key legumes such as *Lotus corniculatus*, supported a higher abundance of chalk grassland moths. Woodcock et al. (2013) showed that the introduction of simple seed mixtures into agriculturally improved grasslands could help support increased diversity of spiders and beetles; and while seed mixtures did not necessarily need to be of the highest diversity to achieve these benefits, the inclusion of legumes did appear to be crucial.

By contrast, Defra project BD5001 (2016) found that the introduction of deep-rooting herbs and legumes had no effect on earthworm biomass, earthworm numbers (Lees et al. 2016) or the foraging success/behaviour of common starlings (*Sturnus vulgaris*). Weisser et al. (2017) also found that positive responses between plant diversity and biodiversity were stronger in above ground taxa rather than below ground.

3.3.1.2 Co-Benefits and Trade-offs

[TOCB Report-3-5B Grassland **EBHE-214C**/**EBHE-214EM**/**ECPW-022**] A field trial on a representative upland farm between 2017 and 2019 showed that reseeding unimproved land as method of improving productivity produced more N_2O emissions per unit of grass yield, than simply applying lime and fertiliser. (Williams et al., 2021). Ploughing and sowing inevitably causes carbon loss from soil.

3.3.1.3 Timescale

Plant species diversity should be improved in the year of implementation (if successful), with associated improvements at higher trophic levels taking longer, particularly where the introduction of diverse swards needs to be aligned with other changes in management as well as other changes at the landscape scale (5-10 years or >10 years). Sowing is not always successful due to e.g., lack of expertise by the farmer relating to soil conditions for effective establishment, poor weather conditions, seed mixes which are inappropriate (to the location or time of sowing) etc.

3.3.1.4 Spatial Issues

Impacts of sward diversity on biodiversity can operate at the sub-field, field, farm or landscape level, depending on the species affected. Connectivity of smaller diverse patches can be critical in producing a larger effect on species richness and/or abundance.

3.3.1.5 Displacement

A reduction in yield from species rich grasslands could result in lower stocking densities in some areas, with displacement of higher stocking densities elsewhere to compensate (Bullock et al. 2011).

3.3.1.6 Maintenance and Longevity

A diverse sward can be short-lived, particularly if it is a seeded herbal ley that is returned to arable. To maintain sward diversity, regular management is required e.g., cutting, grazing and seeding.

3.3.1.7 Climate Adaptation or Mitigation

Species rich grassland contains higher levels of soil carbon than more agriculturally improved grassland but levels of soil carbon will depend on underlying soil type, establishment and time (Norton et al. 2022).

3.3.1.8 Climate Factors

Higher summer temperatures; increased rates of evaporation; less summer rain; decreased soil moisture; an increase in the frequency and severity of droughts. Assessment of grasslands as carbon stores etc.

3.4 Habitat Creation – Ponds/waterbodies and wetlands

3.4.1 ECCA-009; ETPW-062; EBHE-169; EHAZ-082; ECCA-010; EHAZ-049 & EBHE-211

ECCA-009 Create/ restore/ maintain ponds and lakes
ETPW-062 Create/enhance/manage open water habitats
EBHE-169 Restore/ manage ghost ponds
EHAZ-082 Create/ enhance/ manage dry ponds
ECCA-010 Create/ enhance/ maintain washlands
EHAZ-049 Create/enhance/manage freshwater habitats
Also mentioned here is
EBHE-211 *Restore* traditional field ponds, such as dew ponds in calcareous landscapes, using appropriate techniques and materials

Evidence for the effectiveness of the actions to create, restore and maintain ponds, lakes (Moss 2007, Klinge et al. 1998), open waters and freshwater, ghost ponds and traditional field ponds and washlands are drawn from similar sources. An additional commentary is required for ghost ponds (**EBHE-169**) which currently have a geographically restricted evidence base. Further commentary is also provided for washlands (**ECCA-010**), (interpreted here as mostly intermittently flooded land e.g., on floodplains or man-made flood storage areas like the Ouse Washes in Cambridgeshire), which are influenced by the condition of the river environment, and typically dominated by temporary standing waters, which have a more limited ecosystem services evidence base.

For brevity the word ‘pond’ is used for the actions except where the full name of the action is needed for clarity, or when referring to the washlands action.

3.4.1.1 Causality

3.4.1.1.1 Biodiversity adaptation - maintaining / enhancing biodiversity under a changing climate

Actions **ECCA-009**, **ETPW-062**, **EHAZ-049**, **EHAZ-082**, **ECCA-010** have limited evidence in terms of studies specifically addressing responses to climate change but positive outcomes are predicted, consistent with accepted logic chains. All can have major benefits. They are coded (**amber L*****).

Action **EBHE-211**, if focused on traditional waterbodies such as dew ponds, has more limited potential because such sites are uncommon, and are restricted geographically to the specific context of chalk and limestone uplands. For this reason, they are coded (**amber L****). All other farmland ponds, which do not need creation using impermeable liners to retain water (the vast majority of farmland ponds) are covered by the evidence for **ECCA-009**, **ETPW-062**, **EHAZ-049**, **EHAZ-082**, **ECCA-010**.

Ghost ponds (Action **EBHE-169**) also have a limited evidence base and the concept is relatively recently introduced but positive predictions under changing climates are consistent with the evidence logic chain. The evidence so far is that restoring/managing ghost ponds can have major benefits.

There are multiple evidence sources indicating that the creation, restoration and management of ponds may have major benefits for **maintaining / enhancing biodiversity under a changing climate** with a number of lines of evidence suggesting strongly that ponds may help mitigate biodiversity impacts because:

- There is consistent evidence that ponds are the most species-rich freshwater habitats at landscape scale (compared to rivers, lakes, ditches, streams) in all landscapes (summarised in Biggs et al., 2017; original data sources in Williams et al, 2004 and subsequent studies).
- Creating new clean water ponds and managing/restoring ponds in areas where water quality is good, can rapidly add new high-quality habitat (i.e., Priority habitat status) in many different landscape types (Williams et al., 2022).
- New ponds, and to some extent restored ponds where water quality is good, are able to function as key 'stepping stones' in the landscape (Rannap et al., 2009). This is important for allowing biota to respond to climate heating impacts, maintaining meta-populations and reducing the potential for extinction debt (Williams et al., 2020).
- To support this, there is clear evidence from many studies, backed up by a broad swathe of disparate natural history observations, demonstrating the very wide range of freshwater organisms that can quickly disperse to new ponds (Friday, 1988; Williams et al, 2007; Williams, 2017, and summary in Biggs and Williams, 2022).

Additionally, there is recent evidence that ponds have strong carbon capture capabilities (likely greater than woodland) so that ponds may add usefully to carbon stores. However, evidence for this comes from a relatively small number of studies so far (Taylor et al, 2019; Gilbert et al. 2021; Greg et al., 2021).

3.4.1.1.2 Atmospheric deposition of N and exceedance of critical loads.

- Impacts of atmospheric deposition of N and exceedance of critical loads on pond biodiversity is covered elsewhere.

3.4.1.1.3 Connectivity of small 'feature' habitats

Actions **ECCA-009**, **ETPW-062**, **EHAZ-049**, **EHAZ-082**, **ECCA-010** are well tested at multiple sites with outcomes consistent with accepted logic chains and can have major benefits. They are coded (**green***). Action **EBHE-169**, ghost ponds, is also coded (**green***) because for this action the same evidence of contributing to connectivity is applicable as for other actions. Washlands are not small features so have been coded (**N**).

There is a strong overlap in the waterbody types in which freshwater species live. For example, around 80% of wetland plants and 45% of freshwater invertebrate species found in a typical lowland farmed landscapes can use both pond and river habitats (Williams et al., 2004, Biggs et al, 2016). Not surprisingly, therefore, there is evidence of strong connective links both between ponds themselves, and between ponds and other standing and running freshwater habitats:

- Ponds close to other ponds (i.e., occurring in pond and wetland clusters, rather than being more isolated) have been shown to be significantly richer in species and to support more uncommon species in both agricultural landscapes and in protected areas such as SSSIs (Williams et al., 2010; Williams 2018).

- There is also some evidence to suggest a wider effect, with other waterbody types including streams and ditches and lakes benefitting from the presence of ponds in the landscape. For example, in Italy, a lake with numerous ponds nearby recolonised with more diverse algal communities following a period of drying out than did a lake with fewer nearby ponds (Naselli-Flores et al., 2016). In England a range of grey literature indicates that streams and ditches were richer where they were either located close to high quality pond clusters or downstream of on-stream ponds (e.g., Williams 2019, 2021).
- There is also emerging evidence from modelling supporting these observations indicating that the presence of ponds in the landscapes generally increases freshwater species richness at landscape scale nationally in the UK (Borthagaray et al., in prep).

The evidence for the operation of networks of freshwater habitats is reviewed in more detail in Sayer (2014), Biggs et al. (2017) and Biggs and Williams (2022).

3.4.1.1.4 Enhance condition of agricultural land

Actions **ECCA-009**, **ETPW-062**, **EHAZ-049**, **EHAZ-082**, **ECCA-010** are well tested at multiple sites with outcomes consistent with accepted logic chains and can have major benefits. They are coded (**green*****).

Actions **EBHE-169**, ghost ponds, and **ECCA-010** are also coded (**green*****) because the evidence below is also applicable.

There is extensive evidence showing that:

- Ponds are the most species-rich fresh waterbody type in farmland typically supporting 70-80% of freshwater species at catchment level and considerably more rare and threatened species than rivers, lakes, streams or ditches (originally shown in Williams et al., 2004 and other subsequent work including Davies et al., 2008a). This pattern has been replicated in continental Europe (e.g., de Bie et al., 2007); recent Chinese data shows same pattern (Sun et al., 2022).

Their high relative value provides a strong argument that small waters are the most critical freshwater element in most agricultural landscapes, and that changes to ponds, either negative, though loss and neglect, or positive through creation and enhancement, can have profound effects on landscape level freshwater biodiversity (Biggs et al., 2017; Williams et al., 2020). This has recently been shown by Williams et al. (2020), where clean water pond creation resulted in a 26% increase in the number of freshwater plant species, and a 181% increase in rare plant species, in a 10 km2 agricultural catchment. This was set against marked background losses in species of 1% of species and 3% of rare species per year in the catchment and its control. Similarly, Sayer et al. (2012) found significantly greater diversity in managed and restored ponds than those that were unmanaged and in late successional stages.

It is clear from the literature that, when done well, pond creation and management can bring exceptional biodiversity benefits. High quality pond creation has been shown to be particularly valuable. However, the literature also suggests a range of factors which are important for ensuring that creation and management are effective. This includes:

- Water quality: clean water is important for ponds to remain in good condition. Stream- fed ponds tend to be of lower quality because streams typically bring in pollutants and encourage rapid build-up of sediments. As a result, pond creation, is best undertaken using surface-water or groundwater sources, with no links to inflow streams, ditches or drains (Williams et al., 1996, 1998; Williams, 2020).
- Grazing (light to moderate) is generally a positive attribute. Fencing is generally not needed unless persistent heavy grazing is an issue (Williams 2018; 2020).

- Proximity to other wetlands (positive influence): there is a range of evidence showing a network effect due to proximity of ponds to other freshwater habitats (e.g., evidence at national scale in Countryside Survey 2007; Williams et al., 2010).
- Tree shade (related to grazing): moderate shade can have a positive impact on ponds; heavy shade can be detrimental, especially when shade levels have increased recently, and where there are already many shaded ponds in a landscape (Williams 2018; 2020).
- Heavy fish impact: ponds on farmed land heavily stocked with fish often provide very limited biodiversity benefits, though natural fish populations (e.g., in floodplain ponds) create a 'natural' pond habitat type and some species like crucian carp coexist well with other pond biota (Stefanoudis, 2017; Harper et al. 2021). Biggs and Williams (2022) review in more detail the overall impact of fish on ponds and note that c50% of all freshwater biota co-exist with natural fish populations. Continental European work provides more detailed analysis of 'fish stocking' effects, supporting the many anecdotal reports of their detrimental impacts (e.g., Lemmens et al., 2015).
- Dog impacts: although seemingly trivial at first sight, increasing numbers of ponds are suspected to be affected by recreational visitors with dogs and there is strong circumstantial evidence of negative impacts, but no published data and no specific experimental studies are currently available. There are examples of dog impacts in nature reserves where their destructive effects on rare wetland plants are easily detected. However, these are so far only written up in grey literature (e.g., Lansdown and McVeigh, 2019). For ponds near to public rights of way, particularly close to urban areas, there is a local risk that pond, and other small waterbody projects will be compromised by this recreational pressure.
- Duck feeding/stocking: it is widely agreed that stocking waterfowl or feeding them, is damaging to pond ecosystems, but little published information supporting this clearly visible problem. A summary of the management issues is provided in Williams (2020).
- There is evidence that a pond's surrounds have an important impact on pond quality and biodiversity, with ponds bordered by intensive land uses generally less biodiverse than ponds in semi-natural habitats (Williams et al. 1997; Biggs et al., 2017, Morris et al., 2022). Unfortunately, however, there is currently no empirical evidence to suggest the effectiveness of semi-natural buffer zones (of any width) around ponds.

3.4.1.1.5 Enhance condition of semi-natural habitat

All actions except **EBHE-169**, ghost ponds, are well-tested at multiple sites with outcomes consistent with accepted evidence logic chains and are (**green*****).

EBHE-169, ghost ponds, have the potential to bring major benefits but, because they have so far been applied in a relatively small part of England (Norfolk), they are scored (**amber LT*****).

The evidence case for ponds, other freshwaters and washlands in terms of impacts on semi-natural habitats, is broadly underpinned by the same sources as for agricultural land (above). However, the generally better water quality in semi-natural landscapes, and lower levels of other disturbance factors, strengthen the case for the role that ponds pay in these landscapes. Specifically:

- There is good evidence that the highest quality ponds, including the widest range of rare and threatened species, occur in ponds in semi-natural areas (Biggs et al., 2000).
- This includes temporary ponds which, although they support fewer species than permanent ponds, often support uncommon and vulnerable species, particularly in semi-natural landscapes (Collinson et al, 1995, Nicolet et al. 2004; Ewald et al., 2010). For example, 25% of temporary ponds in semi-natural landscapes supported Red Data Book or other uncommon species (Nicolet et al., 2004).

3.4.1.1.6 Favourable condition of SSSIs

All actions except **EBHE-169**, ghost ponds, have been tested at multiple SSSI sites with outcomes consistent with accepted evidence logic chains and are (**green***)**.

EBHE-169, ghost ponds, are likely to have a major benefit on the condition of SSSI. However, at present they have not been created and managed in SSSIs (although they do exist in them), so they are coded (**amber LT*****).

There are estimated to be over 16,400 ponds in England's SSSI's of which around 5% are notified and monitored (Williams, in prep). Many of the best ponds are on SSSI's (Williams 2018), but only around 40% are currently believed to be in Favourable Condition and there is evidence of significant declines in the quality of SSSI ponds (Williams, in prep). For example, Williams (2018) found that in the last 25 years species have been lost from more than two thirds of ponds in protected landscapes, the majority SSSIs, with loss of uncommon species. This decline was thought to be due in part to increasing in shading of ponds and declines in traditional grazing. There was also a negative relationship between pond quality and nutrient levels.

Pond creation on SSSI's has sometimes been shown to bring some exceptional results including the recovery of species that were threatened or believed extinct in semi-natural landscapes (Williams et al., 1997, 2007). The effect of management around ponds has also been generally positive, particularly the removal of secondary woodland around ponds that were traditionally grazed, such as the Thompson Common pingos in Norfolk (Hammond, 2016). There are occasional anecdotal and unpublished stories of damage to high quality ponds on SSSI's through unconsidered management of sites with rare species. Ecological pre-surveys of sites to be managed in high quality landscapes is always advisable. However, these appear to be relatively trivial issues compared to evidence of widespread on-going biodiversity losses due to lack of management as described by Williams (2018). As a result, measures applied to ponds could have a benefit to the condition of the SSSI**.**

3.4.1.1.7 Maintain good condition agricultural land

All actions, except **EBHE-169** ghost ponds and **EBHE-211** traditional field ponds, are well-tested at multiple SSSI sites with outcomes consistent with accepted evidence logic chains and are coded (**green*****).

EBHE-169, ghost ponds, are likely to have a major benefit in maintaining good condition of agricultural land. However, at present they have a geographically limited evidence base, mostly been developed and tested in parts of East Anglia, especially Norfolk, so are coded (**amber LT*****).

EBHE-211 Traditional field ponds created by artificial lining (e.g. dew ponds) are often of limited benefit because they are rare and are likely to lose value quickly as liners fail or are damaged. For this reason, they are coded (**amber LT***).

See 3.4.1.1.4 above for evidence.

3.4.1.1.8 Maintain good condition of semi-natural habitat

All actions except **EBHE-169**, ghost ponds and **EBHE-211**, traditional field ponds, are well-tested at multiple SSSI sites with outcomes consistent with accepted evidence logic chains and are coded (**green*****).

EBHE-169, ghost ponds, are likely to have major benefits in maintaining good condition of agricultural land. However, at present they have a geographically limited evidence base, mostly having been developed and tested in parts of East Anglia, especially Norfolk, so are coded (**amber LT*****).

Traditional field ponds created by artificial lining (e.g., dew ponds) are of limited benefit because they are rare and are likely to lose value quickly as liners degrade or are damaged. For this reason, they are coded (**amber LT***).

See 3.4.1.1.5 above for evidence.

3.4.1.1.9 Presence of rare (red list) species; Presence of priority species

All actions, except **EBHE-169** ghost ponds and **EBHE-211** traditional field ponds, are well-tested with outcomes consistent with accepted evidence logic chains and are coded (**green*****).

EBHE-169, ghost ponds, are likely to have a major benefit maintaining population of rare and priority species. However, at present they have a geographically limited evidence base, mostly having been developed and tested in parts of East Anglia, especially Norfolk, so are coded (**amber L*****).

Traditional field ponds created by artificial lining (e.g., dew ponds), because mainly restricted to farmland and typically isolated in dry landscapes, are vulnerable to (a) pollution and (b) rare species suffering chance extinctions with less chance for recolonisation from other sites as part of a metapopulation. They are therefore coded (**green****) as actions that can have a moderate benefit if well done.

There is good evidence that:

- Ponds support a high proportion of Red List and Priority species; roughly 10% of *all* terrestrial and aquatic England Priority species are recorded in (though not all restricted to) ponds (Freshwater Habitats Trust, 2012).
- There are numerous examples of uncommon or protected species found in ponds although these examples have not been well-documented outside the reports of the Freshwater Habitats Trust. Examples include Britain's most endangered water snail, the Glutinous Snail (Whitfield et al., 1998), now extinct in England following loss from its final pond site, the Natterjack Toad (Beebee et al., 2014), the Tadpole Shrimp known from only two groups of ponds in the UK (Feber et al, 2011) and one of Britain's most endangered water beetles, the Critically Endangered *Haliplus furcatus* (Collinson et al. 1993).
- Ponds are at the cutting edge of new monitoring methods with great crested newt monitoring with environmental DNA now running for seven years since 2015 (Biggs et al., 2015).
- There is growing evidence of the role of ponds for endangered farmland and other birds although studies are limited at present because most work on birds focusses on large sites. Biggs and Williams (2022) note the surprisingly large range of endangered birds using ponds and farmland pond's role has been specifically examined in England by Davies et al. (2016) and Lewis-Phillips et al. (2019a, b, 2020).

3.4.1.1.10 National species occurrence – Invasive Non-Native Species

All actions except washlands are interpreted as having an influence on populations of invasive non-native species. Although all freshwaters can potentially facilitate the spread of alien species, there is good evidence that the risk associated with actions on ponds and small standing waters is generally low and does not compromise the substantial well-evidenced benefits. Field evidence from the Countryside Survey (Williams et al. 2010) showed surprisingly low rates of occurrence of alien species in ponds, and well-documented and well-managed sites have shown little spread over long periods (10-25 years). Generally, alien species are less problematic in high quality freshwaters where they often co-exist with species-rich natural assemblages. This action is coded (**amber*****) in the light of the evidence noted below.

Washlands and larger freshwaters *are* more exposed to river networks which is a major route through the landscape for alien freshwater species. In these locations, spread of non-native species (by birds, people and floodwater) is harder to control so has been coded (**amber D****), indicating some dis-benefits because of the limited ability to control the spread of non-native species in these areas.

There is good evidence that actions to create and manage ponds have had little tendency to spread alien species. There is some evidence that deliberate introductions by people have played a major part in alien plant and animal species spread (Copp et al., 2017; Chan et al., 2019), and limited evidence that this has been exacerbated by natural processes (e.g., animal transport, floods).

Specifically:

- Countryside Survey 2007 (Williams et al. 2020) provides the best evidence for Great Britain of pond infestation by Invasive Non-native Species. 10% of sites were found to have with 1 or more alien plant species characterised as invasive (including *Elodea* and *Impatiens* species). However, in most cases these plants were not dominant at ponds, and there appeared to be no evidence that pond quality was typically degraded by their occurrence (plant richness was significantly greater in ponds with alien taxa). In contrast, pollution and shading were found to have a widespread measurable impact on pond quality.
- Between 1996 and 2007, the Countryside Survey also showed that there was no change in the proportion of ponds with alien species (Williams et al., 2020).
- There is considerable anecdotal evidence that one invasive plant species, *Crassula helmsii,* can significantly degrade ponds, and in some cause local extinctions of uncommon species. However, this is not a universal finding (for example, Ewald 2014).
- There is little evidence to suggest that good pond creation and management results in increased frequency of alien species, if there are stipulations that no plants are deliberately introduced. The Water Friendly Farming project which incorporates Britain's most intensively evaluated pond creation and monitoring programme has, in the 10 years since the project started to create and manage c.50 ponds, has seen no increase in the frequency of alien wetland plant species (Williams et., 2020).

3.4.1.1.11 Evidence of outbreaks of pests and disease

All actions are coded (**amber D***).
Small ponds, pools and seasonal wetlands such as might occur on washlands, are well known as the habitat of the liver fluke intermediate host, the dwarf pond snail, *Lymnaea truncatula*. There is very little published information on the implications of the disease or the host on pond actions in agri-environment schemes. In practice, the impacts on farming systems where stock are managed is minimal and fluke control is a standard agricultural practice.

3.4.1.1.12 Increased abundance, distribution & species richness of pollinators & seed dispersers

All actions are coded (**amber L*****). It is likely that actions involving ponds will have positive benefits on this ES indicator. However, detailed investigations of the use of ponds by pollinators have only begun relatively recently so the evidence base is still quite restricted.

Evidence provided by Walton et al. (2021a, b) and Lewis-Phillips et al. (2020) from sites in East Anglia provides strong support for increasing pollinator diversity around managed ponds. However, almost nothing is known about established farmland or semi-natural pond pollinator assemblages, although much anecdotal natural history evidence suggests it will be substantial.

3.4.1.2 Co-Benefits and Trade-offs

[TOCB Report-3-5D Systems **EBHE-211**] There is good evidence for strong local biodiversity benefits of permanent pond restoration (see **ECCA-009**; Sayer et al. (2012), Lewis-Phillips et al. (2019), Alderton et al 2019). There are likely to be smaller benefits of temporary dew ponds for most groups, but more research is needed in these systems (Alderton et al 2019). Amphibians may do better in temporary ponds because of lack of fish establishment and some species have been found to respond better to dew pond restoration in chalk grassland than others, reflecting colonisation efficiency (anurans were affected more positively than urodeles: Beebee 1997).

3.4.1.3 Magnitude

Typical English lowland farmed countryside ponds:

- Support two thirds of UK freshwater plant and animal species (Williams et al., 2004; Brown et al., 2006; Williams et al. 2020). Note that this does not mean that these species are *only* found in ponds because there is considerable overlap in species' occupancy of different freshwater habitats, and only a minority of species are specifically restricted to running water or standing water. The authors also have unpublished evidence that this pattern occurs in the uplands (specifically from surveys of the catchment of the R. Conwy in North Wales).
- Pond creation can; (a) reverse landscape wide declines in freshwater biodiversity and (b) increase freshwater biodiversity measured as wetland plant richness by 25% (Williams et al. 2020). Although this is the first study in the world to show this, the outcome is consistent with many other observations on the importance of ponds at landscape scale, and the rate of colonisation and richness of clean water ponds.

3.4.1.4 Timescale

Our comments on timescale to achieving high nature conservation status (e.g., meeting Priority pond criteria) apply principally to clean water ponds (i.e., ponds which have minimally impaired water chemistry, equivalent to High status under Water Framework Directive).

Overall, new unpolluted ponds can quickly achieve Priority Habitat status if they are fed by clean water and physically well-designed. Proximity to other freshwater habitats with source populations probably aids colonisation. Specifically:

- New ponds colonise quickly if in good condition; poor quality ponds can also colonise quickly but with lower diversity assemblages (Williams, 2017).
- In 5-6 years, ponds can reach the colonisation asymptote, the end of the initial phase of rapid colonisation and then accumulate species slowly, continuously (if good quality ponds).
- Ponds can achieve Priority Habitat status in 5-10 years and potentially even more quickly if colonised by Priority Species (e.g., great crested newts where colonisation in 12 months is common (Newt Conservation Partnership, 2021).
- New high-quality ponds are commonly and quickly colonised by uncommon species of conservation concern although these have not been well-documented in the research literature. Recent examples include northern damselfly (Blyth, 2014), white-faced darter (Benyon and Daguet, 2005) and a range of species recorded by Freshwater Habitats Trust colonising new high-quality ponds created as part of the Million Ponds Project (2008-12), including Priority species lesser water plantain (*Baldellia ranunculoides*), pillwort (*Pilularia globulifera*), water-violet (*Hottonia palustris*), shining ram's-horn (*Segmentina nitida*) and great crested newt (*Triturus cristatus*).
- So far, the oldest ponds created by authors that have been continuously observed to maintain priority status are now 25 years old. The longest persistence of high-quality ponds in England is c.5000-10,000 years i.e., post-glacial sites.

- Polluted ponds initially gain species rapidly but then decline as build-up of pollutants occurs (Williams et al, 2020a).
- Management of ponds in the farmed landscape has generally led to only moderate improvements in site quality because of management. In the Natural England study of the effects of agri-scheme pond management to be published shortly (Morris et al., 2022), pond quality is mainly determined by (a) quality of surrounds, not management, and (b) quality of water, with groundwater usually cleaner so more likely to produce good quality ponds.

3.4.1.5 Spatial Issues

- For new clean water pond creation programmes, it is beneficial both to target ponds, *and* not to target pond creation. Both strategies have advantages, though neither are well described in the scientific literature.

1. Targeting ponds on areas with existing rich assemblages, clean water or isolated populations increases the chances of uncommon or vulnerable species expanding to new sites.
2. Siting new ponds serendipitously allows surprise colonisation and the exploitation of unpredictable events. In this instance, the key requirement is to ensure ponds are filled by clean water.

   For pond management a range of targeting options are desirable to minimise the risk of damage to high value sites, to maximise the chances of success (by choosing the right ponds to manage) and to recognise when not to manage sites. These targeting options are described in Williams (2020).

- There is a range of evidence that pond creation should be targeted on areas where clean catchments can be provided. The small size of pond catchments has been noted as important characteristic for some time (Davies et., 2007) and is an important part of the practical argument for their special role in the landscape. A recent study undertaken for Natural England in 2021 by the authors indicates that the over-riding influence on the success of pond creation, management and restoration options is the location of the pond in a low intensity, water friendly, landscape (Morris et al., 2022).

- Multiple lines of evidence indicate that the location of new and existing ponds plays an important role in determining the quality of the biota. It has been possible to predict potential locations to optimise pond quality, and this mostly demonstrates that semi-natural landscapes (or areas which simulate the low impact nature of semi-natural landscapes) are optimum for pond quality (Davies et al. 2004).

3.4.1.6 Displacement

- There is no evidence that pond restoration or management cause displacement effects. Pond creation requires habitat conversion, but it may be replacing lower quality habitat.
- At present all available data suggests that new clean ponds add a currently scarce habitat to the landscape. Management of existing, often polluted, ponds equally show no evidence of drawing species detrimentally away from existing habitats to the recently modified habitats.

3.4.1.7 Maintenance and Longevity

3.4.1.7.1 Maintenance

Clean water, high quality, ponds do not automatically need management. Ponds do not inevitably 'fill in and become dry land' a long-promoted myth which has little basis in reality (Biggs et al. 1994; Biggs and Williams, 2022). However, optimum conditions for a range of taxa does typically require management which simulates gentle and natural disturbance which reflects ancient natural processes (e.g., trampling by large animals, disturbance by floods). Temporary ponds may exist for hundreds or thousands of years with little intervention (although may facilitate persistence) as regular drying out prevents accumulation of organic matter.

Despite the popularity of pond management there remains little technical information about its effects. Biggs and Williams (2020) summarise the results of previously unpublished observations made by the Freshwater Habitats Trust, some of the effects of which are also described in the practical management handbook 'The Pond Book' (Williams et al., 2020).

The effects of management on overgrown ungrazed field ponds in East Anglia has been investigated in a series of studies by Sayer and colleagues (e.g., Lewis-Phillips et al., 2019, Walton, 2021a, b) showing that occasional (every 5-10 years) physical management of ponds heavily shaded and overgrown by woody vegetation in ungrazed landscapes was once probably common and has now largely lapsed. Management of ponds with dense woody vegetation cover quickly leads to increases in pond richness at site level and increases in the abundance of semi-terrestrial and terrestrial invertebrates using pond vegetation. Results appear to be assisted by ponds in some East Anglian intensively farmed landscapes often being fed by groundwater; in other parts of England, polluted surface water draining from farmland makes pond management much less effective, with ponds quickly reverting to a polluted state post-management (Williams et al., 2020).

3.4.1.7.2 Longevity

Ponds are often characterised as short lived. Although this can be true for stream-fed, small (e.g., treefall pools) or lined pools, equally there are thousands of ponds in the UK landscape that were created by processes occurring at the end of the last Ice Age, e.g., Pingo ponds of Norfolk and Ice-Age ponds in Herefordshire.

3.4.1.8 Climate Adaptation or Mitigation

There is strong evidence that pond creation (particularly) and to a lesser extent restoration and management, will help mitigate climate change impacts (see section 3.4.1.1.1 Biodiversity adaptation - maintaining / enhancing biodiversity under a changing climate).

There is some evidence to suggest that high quality ponds and pond assemblages will be at risk from climate heating. However, this conclusion is currently mainly based on expert judgement as there has been no direct studies of climate heating impacts on ponds in the UK. Creation and management, as noted above, is likely to provide a very valuable tool for stopping and reversing landscape wide losses of freshwater biodiversity associated with climate change.

There is good evidence that climate heating will increase eutrophication effects in freshwaters, including ponds, although results are not based on direct observation of UK ponds (Moss et al., 2011). Additionally, Ewald (2008) examined the potential impact of climate heating on existing high quality temporary ponds in the New Forest, an extensive semi-natural landscape in southern England. She found that pond communities differed in their response to likely climate heating changes, depending on the stability of the environment and the traits of species within the community. Exposed grassland ponds which contained many obligate temporary pond species showed marked changes in community composition both within and between years. Sheltered woodland sites which contained more generalist species were less disturbed and showed little response to changes in climatic conditions. Together these observations indicate both that climate heating will affect ponds, probably unpredictably depending on the inherent heterogeneity of ponds.

3.4.1.9 Climate Factors / Constraints

High quality, clean-water, pond creation is likely to contribute to mitigation and adaptation of climate heating on freshwater biodiversity, as noted above. Carbon sequestration rates in new ponds have been shown to be substantial (see section 3.4.1.1.1 Biodiversity adaptation above) but there is no information about the stores of carbon found in existing ponds which may be earmarked for restoration and management. New evidence is likely to become available through the current (2021-2024) Horizon 2020 PONDERFUL project which is investigation the role of ponds as nature-based solutions for mitigating climate change impacts.

There is some evidence that polluted ponds, both in urban and rural situations, are a significant source of greenhouse gases including methane, although these data are from North America and Sweden (Holgerson et al. 2016; Peacock et al., 2021). The importance in England's landscape of small waters as greenhouse gas sources has not yet been assessed or compared to other natural and anthropogenic sources. Emerging results from the PONDERFUL project, also indicate a high degree of variability.[3]

There are some constraints due to soil types and geology on the use of ponds as a biodiversity management and climate mitigation tool. However, in a large proportion of England's farmed landscapes, ponds are easily created and despite long periods of losses in the 19th and early 20th century, remain abundant and are now increasing in number (Williams et al., 2010).

3.4.1.10 Benefits and Trade-offs to Farmer/Land manager

The benefits and trade-off for farmers and land managers of ponds and freshwater habitat creation and investigation are not well researched. However practical experience indicates that creating, restoring and managing ponds and freshwater wetland is one of the most popular farmland activities for nature conservation, probably because of high perceived value and generally low land take.

3.4.1.11 Uptake

There are no major barriers to uptake of work on ponds and other wetlands, which are very popular. There is circumstantial evidence from practitioner experience that ponds, and other freshwater habitat work can be hampered by administrative burdens imposed by planning, land use, flood management and other regulations. Though much of this is important and necessary, some streamlining would be beneficial.

Occasionally these issues probably create barriers for some landowners, and may encourage undesirable habitat creation (e.g., digging up an existing valuable wetland habitat to make a pond) and damaging management work resulting from hasty and unprepared projects with too little assessment of existing biological interest. Both problems are real, although for obvious reasons not well-documented. Both can be avoided with standard publicly available guidance information (Freshwater Habitats Trust, 2012; Williams, 2020), much of which is free.

3.4.1.12 Other Notes

This section briefly notes some special features of three of the Actions which modify the evidence interpretations of their ability to deliver the Ecosystems Services under consideration.

Ghost ponds

- So far limited evidence, from one part of the country (essentially, one PhD, one paper).
- Probably groundwater fed which has over-riding positive influence.

[3] https://ponderful.eu/sharing-early-ponderful-research-results-at-intecol-2022/

- In agricultural landscapes constrained by some stressors: at risk of pollution, isolation, sub-optimal management.
- In more semi-natural landscapes should be able to provide the full spectrum of pond benefits.

Evidence on the specific features of ghost ponds is contained in: Walton et al. 2021a, b.
There is a range of evidence to indicate that plants 'recovered' from historic sediments may emerge into a modern environment that is now too polluted for them to survive in in many farmed landscapes (Walton, 2021c.). This problem was previously recognised in the conservation of the endangered wetland pant starfruit (*Damasonium alisma*).

'Traditional' field ponds

- Benefits constrained by location; if fed by clean water, can provide full spectrum of benefits.
- Many 'traditional' field ponds intractably exposed to pollution; in Natural England agri-environment scheme pond survey (Morris et al., 2022) field ponds were equally likely to be poor quality as good.
- Water Friendly Farming evidence shows managing existing polluted ponds brings modest (but some) benefits, substantially less impact than new clear water ponds.
- Dew ponds are often rather poor quality; expensive to make, linings fail so limited lifespan.

Washlands

- A rather rare special case subset of freshwater habitats.
- Limited specific evidence base.
- Effectively a special kind of floodplain.
- Outcome dependent on water quality; may be excellent for comparatively pollution tolerant biota (e.g., water birds, fish).
- Location dependent.

3.5 Habitat Creation - Woodland

3.5.1 ECCM-048; ECPW-044C; ECCA-018; ECPW-071C; EBHE-140C; ECCA-026; EBHE-104 & EBHS-205

ECCM-048	Create woodland on a large scale
ECPW-044C	Create targeted woodland
ECCA-018	Plant or manage large-scale woodland in priority catchments (trade-off – only)
ECPW-071C	Create floodplain woodland
EBHE-140C	Create Ghyll woodland
ECCA-026	Plant a range of native species, including trees grown from locally adapted and genetically diverse seed sources, and from more southerly provenances
EBHE-104	Create a woodland creation plan (trade-off only)
EBHE-205	Create, enhance and manage wood pasture (e.g., through appropriate grazing)

3.5.1.1 Causality

ECCM-048 Large scale woodland creation and **ECCA-018** Plant or managed large-scale woodland in priority catchments are currently proposed primarily to mitigate climate change or for flood protection, yet there are benefits and trade-offs for biodiversity also. The evidence for benefits to biodiversity is (**amber**), the actions are supported, however, there is considerable complexity around their effects, not all taxa benefit and there are some areas where further research is required. It is also not clear if these options are intended to consist of broadleaved woodland creation only, or to also include coniferous woodland, as there are differences in impacts.

Many elements of biodiversity could be affected by these actions, including protected and priority species which live solely in woodlands, as well as generalist species which use them as part of the landscape matrix and those which actively avoid them, preferring open habitats. With the increased interest in woodland creation there have been many recent evidence reviews (Beauchamp et al. 2020, Beauchamp & Jenkin 2020, Post note 2021, Burton et al. 2018, Staddon et al. 2021). Well considered and appropriate woodland creation can benefit forest species and minimise disbenefits to other habitats (Beauchamp et al. 2020). However, woodland creation can have both positive and negative impacts on biodiversity (Burton et al. 2018, Beauchamp et al. 2020), therefore creating woodland on a large scale requires some consideration of the cost-benefits. Factors influencing the effectiveness of woodland creation in terms of enhancing biodiversity including proximity to seed sources, the species of tree planted, environmental conditions e.g., soil nutrient status, management regime and size (Post note). Linked to size, the ability of species to move between woodlands is dependent on the numbers and sizes of patches and connections (such as shelterbelts/lines of trees and hedges) between them.

Large woodlands incorporate greater environmental heterogeneity, provide more ecological niches and support larger populations (Beauchamp et al. 2020). They provide habitat to woodland specialists but potentially displace other wildlife (Beauchamp & Jenkin 2020). There is likely to be an optimal size at which diversity no longer increases with area (Beauchamp et al. 2020). Biodiversity responses will vary by taxa (e.g., microbes, invertebrates, vascular plants, fungi, lichen, mosses, reptiles, amphibians, mammals, and birds) and by landscape context. Studies tend to focus on birds, plants and invertebrates with less studies on other taxa e.g., mammals, soil biome (Burton et al. 2018). There is a lack of long-term studies monitoring the impacts of afforestation on biodiversity (Spake & Doncaster 2017, Burton et al. 2018). The review by Beauchamp et al. (2020) collates evidence from plants, pollinators, soil microbiome, soil invertebrates, birds and mammals, in more detail than is possible to include here.

Positive effects of woodland creation on biodiversity include provision of niche space for habitat specialists and increased structural diversity providing more physical habitat space for all species. Negative impacts of increasing the area and connectivity of woodland could include the potential for spread of invasive species and pathogens, the presence of new seed sources influencing adjacent habitats, or the provision of habitat not suited to edge species or open species (Beauchamp et al. 2020). Hence a (**red ***) has been used for **ECPW-071C** in relation to enhancing and maintaining the condition of agricultural land. Examples of the kinds of disbenefits include effects on bird species that require open conditions which may be predated by species that use forests for cover. Woodland creation can pose risks to pollinators, especially to species associated with open semi-natural habitats (Beauchamp et al. 2020) so (**amber TD****) has been used for action **ECCM-048**. There is some evidence of the positive effects of woodland creation on mammals in the short term, but this relates to abundance of individuals rather than the number of species (richness). No clear positive effects on woodland mammals were detected when woodland expansion was followed over eight years (Lindenmayer et al. 2008).

Woodland area (and landscape context) is/are particularly important influences on bird usage of woodland patches (Beauchamp et al. 2020). This is partly because woodland bird specialists are likely to need large, contiguous areas of habitat into which sufficient numbers of breeding pairs can fit to sustain a local population, and partly because birds in general are mobile and respond to landscape variation at large spatial scales (e.g., Pickett & Siriwardena 2011). The richness of woodland bird specialists continues to rise at the expense of generalists, for larger woods up to 120ha (Gardner et al. 2020). Therefore, with a very long-term (multi-decade) focus for priority bird species, larger woodlands would be recommended. 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. Larger woodlands are considered to be better particularly for large mammals which are area sensitive and occur at low densities (Volenec and Dobson 2020). Small woodlands support edge species but may not provide sufficient conditions for woodland interior specialists, due to light levels, humidity, and foraging area (Beauchamp et al. 2020).

When creating woodland there could be a short-term loss of biodiversity before colonisation of new woodland habitats has taken place. It is important to consider the value of the underlying habitat to be replaced by woodland and the existing priority and other habitats and species that may be influenced by woodland creation within a landscape. For example, wet grasslands and peatlands are home to rarer habitat specialists (Oxbrough et al., 2006; Wilson et al., 2012) and are sensitive to afforestation. Woodland creation should be prevented on existing biodiverse non-woodland habitats (Post note). However, woodland creation on agricultural land of low habitat value is likely to increase biodiversity locally (Post note) and benefit movement of wildlife across landscapes (Staddon et al. 2021). Although woodland planting may result in a stand of similar age structure, encouraging diversity in woodland structure and variation in tree age class is important for many species, e.g., carabids (Burton et al. 2018) and plants (Beauchamp et al. 2020). Not all plant species thought to be most characteristic of ancient woodlands are strictly shade dependent. Many are associated with better lit gaps and rides (Kimberley et al. 2013; Hermy et al. 1999; Peterken & Game 1984; Brown et al. 2015) and so variation in structure is important. The availability of young woodland is important for several bird species (Burgess et al., 2015), but closed canopy sitka spruce plantation is associated with low species diversity, of ground flora particularly (Burton et al. 2018). While woodland creation may benefit pollinators, it may be important to consider whether it does so over and above the creation of non-woodland semi-natural habitats on improved land (Alison et al. 2017, 2016).

Action **ECCA-026 -** Planting a range of native species, including trees grown from locally adapted and genetically diverse seed sources, and from more southerly provenances is designed to enhance the resilience of planted woodland through adaptations to local conditions and to future conditions (with currently more southerly climates) and responds to a variety of literature on the potential benefits of such approaches. The likely advantages of sourcing from local provenance are generally supported (Whittet et al. 2016), although evidence is limited. Even less evidence is available to support sourcing from more southerly provenances given the multiple factors that impact on successful tree establishment and the potential for adverse impacts, hence overall this option is rated as (**amber LD***). Whittet et al. (2016) state that 'the current paucity of knowledge of forest genetic resources in British populations of native tree species suggest that deviations from sourcing currently adapted planting stock are not uniformly applicable throughout the country and that any change to policy ought to be applied judiciously and only under a restricted set of circumstances.' Recently begun work led by Stephen Cavers under the TREESCAPES programme is seeking to provide some evidence on the best approaches for tree planting.

ECPW-044C - create targeted woodland, differs from the 'creation of large-scale woodland' option with an attention to the size and location of created woodland areas. As for large scale woodland creation, evidence for benefits to biodiversity is (**amber**) the action is supported, however, there is considerable complexity around its effects, not all taxa benefit, benefits will be targeted and there are some areas where further research is required. Models of the spatial arrangements of woodland elements indicate that using spatially targeted woodland creation to fill regional 'bottlenecks' may enable species movement where it is currently restricted by lack of woodland habitat (Hodgson et al., 2011), i.e. increasing ecological connectivity. Work by Burton et al. (2018) and Synes et al. (2015) indicates that it is difficult to accommodate multiple species when targeting woodland creation. Further evidence suggests that if woodland creation is restricted to small areas, to best enhance biodiversity, these should be focused on landscapes which are already relatively well wooded, although this could increase biotic risks (i.e. invasion by rhododendron, deer damage) to existing woodlands, (Spake et al. 2019, 2020). However creation of woodland elsewhere, e.g., intensive agricultural landscapes, can have benefits e.g., for pollinators and birds (Beauchamp et al. 2020).

Creation of floodplain woodland **ECPW-071C** is considered likely to substantially benefit biodiversity,[4,5] including increasing populations for uncommon tree species like Black Poplar *Populus nigra* subsp. *Betulifolia* and providing nursery habitats for fish populations (Kerr and Nisbet 1997). However, in locations where this option has been proposed as part of flood alleviation programmes there has been little take up of options and hence evidence is limited for the biodiversity impacts of this option. Most of the biodiversity components have been coded as (**amber T*****) to reflect the lack of evidence and the necessity to carefully consider context.

EBHE-140C There is clear evidence of the biodiversity value of Ghyll woodlands, e.g., those in the Weald of Kent (Burnside et al. 2006)[6] or upland Ghyll habitats. They provide habitat for native trees, ground flora with high diversity of cryptogrammic plants (Waite et al.2010, [5]) and benefit bird species requiring open woodland including redstarts (*Phoenicurus phoenicurus*), pied flycatchers (*Ficedula hypoleuca*) and wood warblers (Phylloscopus sibilatrix) and upland ghyll woodland is important for black grouse (*Tetrao tetrix*)[7]. Evidence is targeted at restoration of existing woodland which could involve fencing to reduce grazing pressure and encourage natural regeneration as well as planting where seed sources are lacking (e.g., Dufton Ghyll wood[8] and Stanley Ghyll wood[9]). There is little evidence on the creation of Ghyll woodland where none exists, although, it would be presumed that biodiversity would benefit from this, it is less certain how successful it would be, hence evidence for this option is considered to be (**amber**).

Evidence for the effects of create, enhance and manage wood pasture on biodiversity have primarily been scored as (**green*****). Creation may refer to re-establishment of wood pasture that was there previously or on new sites, possibly adjacent to existing wood pasture. Appropriate structural diversity (i.e., tussockiness, scrub cover (including bramble – *Rubus fruticosa*) to shelter and protect tree species will enable natural regeneration (Uytvanck 2007). Appropriate grazing is required to lead to successful establishment of species outside scrub thicket and a time gap may be required before grazing starts, to enable establishment of tree species. Evidence on Conservation Evidence[10] indicates that ensuring protection from grazing is important to ensure the successful establishment of pasture trees, but no evaluations on biodiversity impacts of creation have been found. However, a review by Natural England clearly outlines the biodiversity value of these habitats[11] once established.

In terms of enhancement and management, to prevent scrub encroachment and succession to secondary woodland the area of wood pasture and parkland needs to be managed to maintain the desired structure. Preferably this will be through grazing, but in areas where this is not an option, regular mowing can also achieve the same outcome. Stocking levels need to be managed appropriately to enable a heterogeneous structure with tussocks, scrub and variation in sward height. Ideally there needs to be a diversity of age structure of trees to maintain resilience (Uytvanck 2007). Other management methods used include logging, coppicing, pollarding, haymaking, litter-collection and burning (Bakker and Londo, 1998). The mature and ancient trees found in these habitats are important for many diverse species including rare saproxylic fauna which includes some of the most threatened British invertebrates (Buglife). Appropriate tree management such as pollarding, tree protection and retention of deadwood are required.

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[4] https://theriverstrust.org/our-work/our-projects/woodlands-for-water
[5] https://theriverstrust.org/our-work/trees-for-water
[6] https://cris.brighton.ac.uk/ws/portalfiles/portal/4752179/Andrew+Flint+PhD+2014+CD+Rom+version+-+scanned+signature.pdf
[7] https://cieem.net/wp-content/uploads/2019/07/Restoring-ghyll-woodlands-Flora-locale-advisory-note.pdf
[8] https://www.woodlandtrust.org.uk/visiting-woods/woods/dufton-ghyll-wood/
[9] https://www.lakedistrict.gov.uk/caringfor/projects/stanley-ghyll
[10] https://www.conservationevidence.com/actions/644
[11] http://publications.naturalengland.org.uk/publication/4864081829822464

3.5.1.2 Co-Benefits and Trade-offs

[TOCB Report-3-6 Carbon] Ghyll woodlands are typically linear woody features, found on steep valley slopes and are composed of diverse canopy forming species and a diverse understory (Burnside et al. 2006). A study of ghyll woodlands in the high Weald found that the most common woodland types were *Quercus robur/Pteridium aquilinum/ Rubus fruticosus* woodland and *Fraxinus excelsior/Acer campestre/Mercurialis perennis* woodland. No evidence for the carbon sequestration and storage potential of ghyll woodlands specifically could be found for this review. However, positive effects of targeted woodland creation and the creation of woody linear features more generally for carbon sequestration and storage can be found in Report-3-6 *Carbon*, sections on 'Habitat Creation – Woodland' and on 'Woody Features', which provide an approximate estimate for the impact of ghyll woodland creation on carbon storage.

[TOCB Report-3 Soils] Creation of Ghyll woodland is likely to have positive benefits for soil health as with woodland on other land but it is unlikely to result in reductions in nutrient input as any Ghyll land use tends to be very extensive due to steep slopes and thin soils.

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	LTD*

3.5.1.3 Magnitude

As stated above, creation of woodland has mixed effects on biodiversity dependent on multiple factors.

3.5.1.4 Timescale

Relevant to all woodland creation actions (**ECCM-048**, **ECPW-044C**, **ECCA-018**, **ECPW-071C** and **EBHE-140C**), species respond to conservation action in different ways, and temporal lags in species response could mask the ability to observe progress towards conservation success (Watts et al. 2020). The timescale for the establishment of species, including plant communities, varies depending upon soil nutrient levels, climatic region, disturbance, potential for species dispersal, dispersal mechanism, distance to existing woodland and connectivity of woody linear features such as hedgerows. There is conflicting evidence about the timescales for plant community assembly after woodland creation, e.g., vascular plants show different responses to bryophytes. Many impacts are long term and so can take longer than the duration of most studies making it difficult to get evidence. Plant community biodiversity and other benefits could begin to appear in years 0-5 after woodland establishment and continue to develop over many years as the trees mature (ER4-Keenleyside et al. 2019). Restoration of full canopy cover from grassland could take 20-30 years but then transition from a flora of light demanding species to shade tolerant up to 40 years (Harmer et al. 2001). Even then, although established woodland plant communities may include understorey and canopy specialists, poorly dispersing and rare species are likely to remain absent for longer especially where legacy effects of disturbance and increased soil nutrients persist (Dupouey et al. 2002; Strengbom et al. 2001; Naaf and Kolk 2015). Another study showed that it took 70-80 years where new planting was adjacent to existing ancient woodland for species richness and composition to become comparable to the existing woodland (Kimberley et al. 2014).

For birds, time lags, especially for specialists, are very long, so a focus of multiple decades is required, opening a potential issue of unknown interactions with climate change. Farm woods of around ten years old attract scrub, hedgerow and open-country 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).

The impacts of forest planting on invertebrate diversity will also depend on the time since planting (i.e., woodland maturity). There are likely to be short-term impacts (i.e., after several years) on populations of species already present. It is over a decadal timeframe, however, that significant changes in soil invertebrate biodiversity are likely to take place, as species are able to migrate into new habitat. The length of time that it takes to evidence significant change in soil invertebrates is also partly determined via changes in soil properties. For example, earthworm diversity is also controlled by inherent soil properties such as pH and texture.

3.5.1.5 Spatial Issues

Newly planted woodland (including targeted woodland) or increases in extent of existing woodland encouraged by natural expansion in the absence of grazing, should focus on augmenting existing long-continuity woodlands (Beauchamp et al. 2020) although conditions in the buffering woodland ought to be similar. If soil conditions reflect agricultural legacy and are very different in pH, macronutrient levels and seedbank composition then these are likely to make establishment more difficult (Kimberley et al. 2014; Govaert et al. 2020). 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. Adding woodland cover can under certain contexts increase risk of invasion by rhododendron (Spake et al. 2019) and risk of deer damage (Spake et al. 2020).

There is limited evidence on the nuances of where woodland creation or management are best placed to benefit pollinators, although there is clear evidence that effects of woodland on crop pollination are distance-dependent.

Creation of floodplain woodland **ECPW-071C** would inevitably need to be on floodplains, however low-lying land is often of high agricultural quality or developed. Often floodplains have become disconnected from the watercourse and there may need to be restoration of natural geomorphological processes.

3.5.1.6 Displacement

Woodland created on current agriculturally improved land results in a displacement of production of either livestock or arable cropping. This is not the case for Ghyll woodland **EBHE-140C**.

3.5.1.7 Maintenance and Longevity

Woodland creation has both short and very long-term impacts on biodiversity but because revenue comes many years after planting, adequate funding and support for maintenance and, where necessary, management of the woodland resource is required. In the establishment phases, careful management may be needed.

3.5.1.8 Climate Adaptation or Mitigation

Woodland creation contributes to C sequestration (in biomass, soils and harvested forest products). It also contributes to adaptation or mitigation within the habitat by providing canopy cover that will influence temperatures and microclimate and provide shelter to shade tolerant species (Thomas et al. 2016)[12]. The microclimatic effects may also impact surrounding land and running water.

[12] https://environmentalevidence.org/project/what-are-the-effects-of-wooded-riparian-zones-on-stream-temperature-systematic-review/

Ensuring good condition of wood pasture (**EBHE-205**) through regeneration and replanting to establish new generations of trees to replace individuals and species that are lost or likely to be lost under climate change will be a beneficial adaptation.

3.5.1.9 Climate Factors / Constraints

There may be constraints on the species and genetic provenance of species in future e.g., beech, see action **ECCA-026** under 3.5.1.1.

There are likely to be issues with a lack of sufficient nursery stock to enable very large scale tree planting in England [13].

EBHE-205 Evidence suggests that beech dominated wood pasture in the south of England will be increasingly vulnerable to drought, particularly on freely-draining soils and soils subject to seasonal water-logging. More generally, drought and an increased frequency of storms pose a threat to veteran trees, which are a distinctive feature of much wood pasture and parkland (NE and RSPB 2019). Hotter and warmer winters could result in increases of pests (Read 2009).

Benefits and Trade-offs to Farmer/land manager

Decisions about whether, when and where to create woodlands are influenced in part by the benefits which they provide, often multiple benefits (Beauchamp & Jenkin 2020) and the extent to which they displace food production. The main drivers for many landowners are economic, income generation, long term investment or tax relief. Commercial production may be a key route to generating revenue, including wood fuel, recreation income and payments for carbon sequestration through grants. Indirect benefits include improvements in air and water quality, the water holding capacity of land, local climate regulation (shelter/shade) and, in some areas, landscape quality.

3.5.1.10 Uptake

Evidence suggests that incentives need to be consistent with wider management objectives – financial incentives alone won't change management if they don't align with landowner beliefs, values and ambitions for land management. Barriers to woodland creation include grants being insufficient for longer- term management or to overcome the missed opportunity costs resulting from a permanent change of land use, including lost future revenue from productive land. Major barriers to implementation are farmers' lack of knowledge, technical skills and time to manage woodland, and possibly unwillingness to invest capital in non-agricultural land management (Keenleyside et al. 2019). The long timescales of forestry are off-putting to many landowners, with revenue from forestry coming after too long a timeframe. Land value also decreases once woodland is planted. (Beauchamp & Jenkin 2020). There may also be a perception amongst farmers that loss of productive land to woodland is culturally unacceptable, there may also be concerns about undesirable landscape change. Hence, there may be social barriers as well as lack of skills and training. Nisbet and Thomas (2008) show that the creation of floodplain woodland (as relevant to **ECPW-071C**) in the Ripon catchment was hugely constrained by lack of uptake by farmers and landowners.

3.6 Habitat Creation – Woody habitats

3.6.1 EBHE-209 & ECCM-055: Create orchards

EBHE-209	Create, restore, manage traditional orchards with local varieties of fruit tree
ECCM-055	Plant traditional orchards

[13] https://www.nao.org.uk/wp-content/uploads/2022/03/Tree-planting-in-England-Summary.pdf

3.6.1.1 Causality

There is limited evidence for the impacts on biodiversity for **EBHE-209** or **ECCM-055** (**amber L***) Create (or plant) traditional orchards with local varieties of fruit tree, particularly for the use of local varieties. However, a review by NE clearly outlines the biodiversity value of these habitats[14] once established.

Traditional management includes managing grassland by grazing or hay cutting, maintaining the characteristic tree form by pruning, restoring tree numbers, protect trees from damage and keeping standing deadwood and some deadwood on living trees.

3.6.1.2 Co-Benefits and Trade-offs

[TOCB Report-3-6 Carbon **EBHE-209**] Reports of carbon storage and sequestration in English orchards are scarce. A review of carbon storage and sequestration by habitat in the UK estimated that traditional orchards can support soil carbon contents of an average of 73.75 t C ha -1 to a depth of 30cm (with a range of observed values between 47 and 111 t C ha-1).

Vegetation in traditional orchards is estimated to contain an average of 21.4 t C ha-1 (with reported range of 8.6 to 230.4 t C ha-1) (Gregg et al., 2021). Both mean values are associated with a low degree of confidence due to a lack of evidence in the literature (one published study was identified; Robertson et al., 2012). The same study reported average rates of carbon sequestration of -2.89 t CO2eq ha-1 yr-1 (range: -5.89 to +1.65 t CO2eq ha-1 yr-1) in traditionally managed orchards.

Material that is removed from orchards is reportedly often burnt, returning much of the sequestered carbon to the atmosphere. As a result of these outgoing fluxes, orchards have a net sequestration rate (not including the emissions associated with the processing of produce or management practices) smaller than other types of woodland and can result in net carbon emissions (Gregg et al., 2021; Matthews, 2020). Therefore, expert opinion suggests an intensification in management may reduce net sequestration.

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

[TOCB Report-3-3 Soils **EBHE-209EM** & **EBHE-209C**] Enhancing and managing woodlands and tree shelterbelts will secure soil erosion and soil health benefits. Targeted introduction of trees, shrubs and scrub to the agricultural landscape is likely to result in an overall reduction in soil erosion risk and a moderate to major positive benefits to soil quality in terms of improved soil structure and increased soil organic matter.

3.6.1.3 Other Assessments

There is currently insufficient evidence to assess this action in terms of magnitude, timescale, spatial issues, displacement, maintenance and longevity, climate adaptation or mitigation, climate factors/constraints, benefits and trade-offs to farmer/land manager or uptake.

14http://publications.naturalengland.org.uk/publication/19007

3.6.2 ECCM-051C: Create buffer zones around ancient woodland (including through extension of existing woodland).

This option is covered here, but evidence included is relevant to these other actions;

- ECCA-024 Create new areas of habitat adjacent to existing habitat patches to increase patch size and help sustain more viable species populations,
- ECCM-051EM Enhance or manage buffer zones around ancient woodland (including through extension of existing woodland), and
- ECCA-050 Restore degraded areas of habitat adjacent to existing habitat patches to increase patch size and help sustain more viable species populations (currently listed in Restoration, Management and Enhancement).

3.6.2.1 Causality

As with other woodland creation options many of the biodiversity attributes have been coded as (**amber*****, context dependent (**T**) with some disbenefits (**D**)), see **ECCM-048**. Well considered and appropriate woodland creation can benefit forest species and minimise disbenefits to other habitats (Beauchamp et al. 2020). However, woodland creation can have both positive and negative impacts on biodiversity (Burton et al 2018, Beauchamp et al. 2020). It can also increase biotic risks (i.e., invasion by rhododendron, deer damage) to existing woodlands, depending on context (Spake et al. 2019, 2020). Woodland specialists are most likely to benefit where native woodland creation is adjacent to existing ancient woodland (Beauchamp et al. 2020) although it is necessary to consider what habitat is being replaced to create the buffer, if it is agricultural land then there could be displacement but no loss of other semi-natural habitat. Targeting new native woodland adjacent to ancient woodland patches increases core habitat area and functional network size, enabling faster colonisation of woodland species (Burton et al. 2018, Bailey, Lee and Thompson, 2006; Hope, Humphrey and Stone, 2006). It provides locally adapted seeds (POST 2021, Merckx & Pereira 2015). Locating next to an existing woodland provides a greater complexity and diversity of habitats to support species, enabling refuge for species during disturbances and improving resilience. It also provides a buffer for the existing woodland (Beauchamp et al. 2020). However, conditions in the buffering woodland ought to be similar. If soil conditions reflect agricultural legacy and are very different in pH, macronutrient levels and seedbank composition then these are likely to make establishment more difficult (Beauchamp et al. 2020, Kimberley et al. 2014; Govaert et al. 2020). Jacquemyn et al. (2003) 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.

3.6.2.2 Co-Benefits and Trade-offs

[TOCB from Report-3-6 Carbon **ECCA-024**] Creating additional woodland can sequester carbon at the fastest rate of any semi natural habitat (tC ha-1 yr -1) see section on ‘Habitat Creation – Woodland’ of the carbon sequestration review (Report-3-6). In addition, a larger patch size and greater population genetic diversity can potentially increase resilience to pressures on woodland, see Report 3-6.

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

[TOCB Report-3-3 Soils **ECCA-024**] 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-3-6 Carbon **ECCM-015C**] Buffer zones could be composed of a range of habitat types, and as such their effect on carbon sequestration is context dependent. The largest positive effect would be expected from creating additional woody vegetation (see Report-3-6 *Carbon*, section on 'Habitat Creation – Woodland'). Ancient woodland is home to a significant carbon stock and preserving these woodlands will prevent significant future emissions of carbon to the atmosphere (Matthews, 2020). In addition to expanding population gene pools, the creation of buffer zones of sufficient height can also prevent wind damage and disturbance (Poëtte et al., 2017). The creation of other habitats, such as scrub or diverse permanent grassland may also have a positive effect on above and below ground biomass, compared to an arable baseline.

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	TD**

Evidence to assess this action in terms of maintenance and longevity, climate adaptation or mitigation, climate factors/constraints, benefits and trade-offs to farmer/land manager or uptake is covered in the 'HABITAT CREATION – WOODLAND' section of this report.

3.6.2.3 Magnitude

Kimberley et al. (2014) found that 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.

3.6.2.4 Timescale

See HABITAT CREATION - WOODLAND, timescale issues relevant there are also relevant here. Adjacency to existing habitat may speed up the process of species colonisation but it also may be that without this source of species it will not be possible to achieve the same species composition as an ancient woodland.

3.6.2.5 Spatial Issues

These options are explicitly spatial in relation to current woodland.

3.6.2.6 Displacement

Any woodland or woodland buffer created on current agriculturally improved land results in a displacement of production of either livestock or arable cropping. Buffers surrounding ancient woodland that was previously adjacent to intensively used land will be particularly beneficial to biodiversity, through avoided impacts of fertilisers, pesticides and habitats providing limited species niches.

3.6.3 EBHE-104: Create a woodland creation plan

Co-benefits and trade-offs only are assessed here.

3.6.3.1 Co-Benefits and Trade-offs

[TOCB Report-3-5C **EBHE-004**] To mitigate disbenefits from woodland expansion on biodiversity, site-based evaluations are necessary, careful forest design planning and tailored management of new woodland sites. Expert value judgements may be required to establish which elements of biodiversity and ecosystem services are prioritised at both local and national scale. However, these local judgements must sit within in a strategic landscape, regional and national framework to ensure all habitats are conserved (Beauchamp et al. 2020). No evidence was found regarding biodiversity benefits resulting from the creation of a woodland management plan. All plans have been scored as (green T).

[TOCB Report-3-5B Grassland **EBHE-004**] Biodiversity benefits will depend on the existing land cover and management, and on the type of woodland that replaces it, how this is managed. Significant benefits for a range of taxa and species can be expected if the plans follow the principle of maximising habitat value within the woodland and the landscape within which it is placed.

3.6.4 EBHE-203C: Create targeted scrub

3.6.4.1 Causality

There is evidence that creating targeted scrub (**EBHE-203C**) can have positive effects on biodiversity (Mortimer et al. 2000, Day, Symes & Robertson 2006). Scrub can be valuable to many different taxa and is generally considered an important component of many habitats although it also has the potential to invade and spread and lead to successional development damaging early successional habitats so overall evidence is (**amber****) with some potential disbenefits (**D**). It is an important habitat for several breeding and wintering bird species and is used as a safe roost site and a source of invertebrates or berries as food. Many invertebrates feed on shrubs and many more on the associated lichens, algae and fungi of the bark and wood. Scrub also provides sources of food and shelter to mammals e.g., badger, deer, rabbits, foxes, dormice, bats.

Scrub creation is likely to be detrimental to particular plant communities as it is a late successional stage of vegetation and can exclude many species. However, some plant species will exist at the edges of areas of scrub and epiphytic species may also benefit. Scrub creation may be detrimental to reptiles and amphibians e.g., sand lizards and great crested newts, although in general a mosaic of scrub with variation in structure is likely to be beneficial.

Natural regeneration would be the recommended approach to increase scrub provided there is sufficient source material available. Where a relict scrub community remains, ground preparation may be required to remove a dense mat of rank vegetation that is suppressing natural regeneration. Grazing pressure may also need to be reduced to encourage establishment.

3.6.4.2 Co-Benefits and Trade-offs

No evidence assessed.

3.6.4.3 Magnitude

No evidence.

3.6.4.4 Timescale

Natural regeneration of scrub can be very rapid (i.e., within a few years), dependent on location and proximity to existing scrub in hedgerows etc. Successful establishment of scrub by planting is dependent on species choice, suitable soil conditions, adequate protection and water.

3.6.4.5 Spatial issues

Creation of scrub habitats will be most successful adjacent to existing scrubby features.

3.6.4.6 Displacement

Scrub created on current agriculturally improved land results in a displacement of production of either livestock or arable cropping and may lead to succession to woodland habitats in the longer term.

3.6.4.7 Maintenance and Longevity

Maintenance of scrub habitats is required to prevent succession to woodland – can include grazing/browsing, burning and water table management.

3.6.4.8 Climate Adaptation or Mitigation

Scrub creation contributes to C sequestration (in biomass, soils and harvested forest products). It also contributes to adaptation or mitigation within the habitat by providing canopy cover that will influence temperatures and microclimate and provide shelter to shade tolerant species. In wetland habitats scrub encroachment can lead to drying out and lowering of the water table which has implications for climate change adaptation.

3.6.4.9 Climate Factors / Constraints

No evidence.

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

As for woodland.

3.6.4.11 Uptake

Primarily as for woodland.

3.7 Habitat Creation – Woody features

3.7.1 ECAR-033C; ECCM-024 & ECCA-036

ECAR-033C Create tree shelter belts near sensitive habitats
ECCM-024 Plant or manage trees outside of woodlands, including shelterbelts
ECCA-036 Plant trees alongside water courses to provide shade and reduce water temperatures within rivers

3.7.1.1 Causality

Evidence for actions **ECAR-033C** and **ECCM-024** has been scored identically except for impacts on atmospheric deposition. For most services the evidence is scored as (**amber LT* or ****) indicating that there is limited evidence for the impacts of these actions on biodiversity and that effects are likely to be context dependent but positive. These actions will clearly have some of the same benefits as woodland creation (see above) resulting from the beneficial effects of adding trees to a landscape. **ECCA-036** is scored as (**green***) for both biodiversity adaptation and maintain good condition of semi-natural habitats, because of positive effects of tree shading on biodiversity in water (reducing temperatures) (Garner et al. 2015, Thomas et al. 2016, [11]), and on structural diversity in riverside habitats[15].

In general, evidence indicates that the presence of scattered trees outside of woodlands, including shelterbelts can be highly beneficial for biodiversity within landscapes (Prevedello et al. 2017). There is however, limited published evidence on the impacts of shelterbelt creation **ECCM-024** on biodiversity, with most studies focusing on impacts on agriculture. Evidence that does exist includes a study by

[15] https://www.conservationevidence.com/actions/141

Littlejohn et al. (2019) which showed that planted Miscanthus shelterbelts had positive impacts on earthworm numbers and that refuges provided in shelterbelts (for bees and reptiles) were more likely to be occupied than field edges with fences alone. Actions have been coded as (**amber LT*)** for enhance and maintain biodiversity because of a lack of direct evidence relating to shelterbelts.

Evidence indicates that riverine water temperatures are significantly reduced by the presence of trees casting shade over the river (Broadmeadow et al. 2011) (**ECCA-036**), which is beneficial for biodiversity. Evidence is focused on existing trees, there are no studies showing how planting of trees has altered river temperature. There could be potential disbenefits to existing riverine habitats from tree planting and increasing connectivity to allow the spread of invasive species.

3.7.1.2 Co-Benefits and Trade-offs

Shelterbelts may enhance biodiversity in sensitive habitats (**ECAR-033C**) by absorbing air-borne pollutants and reducing wind flow. Studies by Bealey et al. 2016 indicate that tree planting adjacent to sensitive sites could mitigate emissions by 0.14-6%.

3.7.1.3 Magnitude

The effects of shelterbelt creation **ECCM-024** on biodiversity will depend on the extent and nature of existing habitats within the landscape. Effects on species will vary according to species preferences. Broadmeadow et al. 2011 found that the effect of the presence of trees on water bodies **ECCA-036** was sufficient to keep summer water temperatures below lethal levels for fish species.

3.7.1.4 Timescale

Issues are similar to those for woodland creation, depending on establishment time, locality, and species present.

3.7.1.5 Spatial Issues

Proximity of trees outside of woodland and shelterbelts to other wooded habitats and the nature of surrounding habitats will impact on the potential for biodiversity increases.

3.7.1.6 Displacement

Shelterbelts and riverine trees follow linear features and may mean minimal displacement of habitats used for other purposes.

3.7.1.7 Maintenance and Longevity

Kort (1988) presented information suggesting that the payback period for new shelterbelts (in terms of impacts on crops) ranged from 15 to 40 years. Clearly the width of the shelterbelt, or hedge, will have a considerable impact on this, along with the locality, local climate and soil.

3.7.1.8 Climate Adaptation or Mitigation

Tree planting contributes to C sequestration (in biomass and soils). It also contributes to adaptation or mitigation in adjacent habitats by providing canopy cover that will influence temperatures and microclimate and provide shelter to shade tolerant species.

3.7.1.9 Climate Factors / Constraints

No information.

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

Shelterbelts provide a range of services. Whether shelterbelts will have a net positive or negative effect on crop growth through hydrological effects will vary with crop type, hedge type, climate, topography and soils. In Italy, windbreaks were found to reduce water loss from evapotranspiration for a distance of 12.7 times the windbreak height. Water use efficiency (WUE) within this zone was 1.15 compared to 0.70 outside of it (Campi et al. 2009). In Canada, in low rainfall years, crop yields fell immediately adjacent to a shelterbelt due to competition for water but were slightly higher in the next band outwards into the crop (Kowalchuk & Jong 1995). Models of water use efficiency developed in the USA indicate that in some cases shelterbelts could contribute towards greater water use efficiency in crops, but that the exact mechanism by which they do so would be difficult to determine (Davis et al. 1988).

Biber (1988), covering Europe generally, presents figures for increases in yield in different crops that may be expected from planting shelterbelts. These include sugar beet up by 11-12%, wheat 6-26%, maize 10-15%, grass 27-67%, potatoes 9-17%, apples 16-75% and pears 121%. It is not clear whether or not these figures take account of land lost to the shelterbelts and yield reductions due to shading and competition between crops and trees. Other effects of shelterbelts include reducing soil erosion and storing carbon (Wolton et al. 2016).

3.7.1.11 Uptake

No evidence.

3.7.1.12 Other Notes

N/A.

3.8 Actions for Habitats with Specific Hydrological Characteristics: Habitat Creation - Wetlands

3.8.1 EBHE-164C; ECCA-007C; ECCA-013C; EHAZ-129C; ECCM-039 & ECCM-038

EBHE-164C	Create wetland habitats, duplicates the following:
ECCA-007C	Create wetland habitat mosaics, including creating the appropriate hydrological conditions
ECCA-013C	Create artificial wetlands
EHAZ-129C	Create fen
ECCM-039	Restore areas of farmed peatland to wetland
ECCM-038	Raise water levels in areas of farmed peatland and adapt farming systems accordingly

These options have been merged because there is a lot of overlap. Wetland habitats that may be created include fen, reedbeds, most likely to be on land that was previously wetland and has been drained for agriculture. Terminology includes restored and created wetland.

3.8.1.1 Causality

There is good evidence (**green**) that some types of wetlands can be created and will benefit the condition of semi-natural habitats although they are unlikely to be of the same quality as older, undrained sites. For favourable condition of SSSI's and rare species the evidence is scored as (**amber**) and context dependent (**T**). In studies on Wicken Fen where arable sites were drained within a landscape of intense agriculture and older remnant fen even after long time periods the resulting vegetation was different to the target/undrained wetland (Stroh 2012, 2013). Beetle numbers were also low on restored fen possibly due to dispersal constraints or habitat suitability. Increased soil moisture and decreased vegetation height were indicators of higher activity of scarce beetles and could become restoration targets (Martay et al. 2011).

Restoration should be aimed at restoring the wetland system rather than individual sites (Grootjans et al. 2007). In a different study reedbed that had been created from arable land 40 years prior to the study but restored from a dry state 12 years prior to study, it was found that ancient and new sites had similar numbers of invertebrate taxa and community composition, but new sites had fewer individuals and particularly fewer individuals of rare species (Hardman 2010).

3.8.1.2 Co-Benefits and Trade-offs

[TOCB Report-3-5A Croplands **ECCM-038** and others] Fenland peatland was found to have reduced soil carbon loss when managed under conservation grassland or with raised water levels, compared to intensive arable production (Graves and Morris, 2013).

[TOCB Report3-5B Grassland **ECCM-038**] Avoidance of GHG emissions from drained peatland. Water quality. Flood risk management. Natural flood management.

3.8.1.3 Timescale

Creation of fen from arable land found that the oldest site (60 years) had the most similar vegetation to the target but it was not possible to restore the vegetation exactly (Stroh et al 2013). Some wetland/peatland habitats can only be restored over long timescales, into centuries e.g., Raised bogs whereas others such as open water habitats (reedbeds) develop interesting bird communities quite quickly- few years (Ward). RSPB's Lakenheath Fen project took around ten years from inception to bitterns becoming established (Natural England and RSPB 2019). Twelve years of restoration was not sufficient to entirely restore invertebrate communities in terms of abundance, but that all species had been restored by this point (Hardman 2010).

3.8.1.4 Spatial Issues

There are spatial issues with creation of wetland, suitable environmental conditions, and adjacency to existing wetland.

3.8.1.5 Displacement

Creation of wetland on arable land displaces production elsewhere.

3.8.1.6 Maintenance and Longevity

Ongoing management is required to maintain wetland.

3.8.1.7 Climate Adaptation or Mitigation

Wetland creation should be beneficial for C sequestration and climate adaptation.

3.8.1.8 Climate Factors / Constraints

Hydrological management is affected by water tables in surrounding areas which are themselves affected by climate factors and water and land use.

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

Loss of productive arable land. Natural Flood Management.

3.8.1.10 Uptake

No assessment.

3.8.1.11 Other Notes

N/A.

3.9 Actions for Habitats with Specific Hydrological Characteristics: Habitat Management & Enhancement - Wetlands

3.9.1 ECCA-007EM; ECCM-030; EBHA-164EM; EHAZ-063; ECCA-013EM; ECCM-032 & EHAZ-129EM

ECCA-007EM	Enhance/ manage wetland habitat mosaics, including creating the appropriate hydrological conditions (not including grazing, which is covered below)
ECCM-030	Restore/ manage upland and lowland peatlands including blanket bog and raised bog
EBHE-164EM	Enhance/ manage wetland habitats
EHAZ-063	Block drains, ditches and grips
ECCA-013EM	Enhance/ maintain artificial wetlands
ECCM-032	Manage hydrology in wetland habitats to restore functional processes
EHAZ-129EM	Enhance or manage fen

3.9.1.1 Causality

The evidence that these actions will have beneficial effects on the condition of semi-natural habitats is (**green)**. The impact on the Favourable condition of SSSI's and rare species has been coded as (**amber**) and context dependent (******). There is uncertainty about how successful restoration actions might be. This includes whether the hydrology can be successfully restored (**ECCA-007EM**), how long it might take and whether the restored habitat is of the same quality as an intact peatland.

Most of the relevant (historical) literature comes from the opposite action, i.e., the impact of wetland drainage (Eppinga 2009, Talbot 2010, Gatis et al. 2016) and compares drained wetland to undrained. Restoration of appropriate hydrological conditions typically involves raising the water table nearer to the surface and re-establishing peat forming fen or bog vegetation (Moxey, 2011; NE, 2011; Moxey and Moran, 2014). Grip blocking and gully blocking are the most widely applied techniques using peat turves, plastic piles, wooden dams, heather bales, straw bales and stone (Cris et al., 2011, Holden et al. 2008, Shephard 2013) to re-wet the peat and increase water levels. Other techniques may also be required e.g., peat bunding and sluiceways. Grip blocking on large scales is relatively recent and hence long-term studies on its effects are less common and the ecological consequences of such management interventions are poorly understood (Beadle et al. 2015). Wetland responses to altered hydrological regimes have been found to be complex; spatially and temporally heterogeneous, and mediated by feedbacks among vegetation, peat structure and hydrology (Talbot 2010, Williamson et al. 2017, Gatis et al. 2016). Grip blocking has been found to increase the cover of specialist bog plants such as *Eriophorum vaginatum* and *E. angustifolium* (Komulainen et al., 1999) and rich-fen species including Sphagnum and wetland bryophytes that prefer a higher water table. Lindsay (2010) suggested that ditch blocking on blanket bogs will lead to an increase in the abundance of Sphagnum mosses at the expense of cotton grasses. Other studies did not find the expected response of an increase in wet species and decrease in dry species (Green, Williamson 2017, Bellamy 2012) with ditch blocking, which could be related to insufficient changes in the water table, insufficient time taken since blocking or subsidence. Restoring the micro-topography of the bog surface (Forrest 1971, Belyea and Clymo 2001, Eppinga 2009), is an important element in restoring natural function, but may take careful water management and time. The bog surface consists of hummocks, lawns and hollows, with sphagnum and sedges in the hollows and other vascular plants such as *Eriophorum vaginatum* in the hummocks (Eppinga 2009). Laine et al. (2007) found that average water-table depths could differ by 5–10 cm between lawns and hollows and by 10–15 cm between lawns and hummocks. This may mean that changes in mean water table of only a few cm are insufficient to change vegetation composition. Correct maintenance of hydrology is essential for recolonisation of bog species and should be a priority (Carroll et al. 2009). Research on bog pools (Beadle et al. 2015) indicated that aquatic biota would benefit from land managers creating an array of differently sized pools behind the blocked grips and prioritising landscapes close to existing water bodies to encourage faster colonisation.

While re-wetting and re-naturalisation (development of appropriate vegetation) are achievable for degraded blanket bogs, regeneration (renewed accumulation of peat) may be more difficult to achieve in a short timescale.

3.9.1.2 Co-Benefits and Trade-offs

No evidence.

3.9.1.3 Magnitude

These is a lack of specific evidence on the magnitude of biodiversity enhancement as a result of wetland restoration. When re-wetting, plant communities may change whilst overall species richness remains similar. Evidence relating to species richness of stream invertebrates in catchments with ditch blocking showed that the community composition of restored sites was more similar to that of intact, rather than drained, areas (Ranchunder et al. 2011).

3.9.1.4 Timescale

Some peatland habitats, like raised bogs, can only be restored over century long timescales. In contrast, open water habitats can develop interesting bird communities over a few years.

Biodiversity changes following grip blocking may take many years, Bellamy et al. 2012 found that cover of species indicative of bog recovery was greater where the drains had been blocked for the longest time (11 years). A study in north Wales showed that blocking drainage ditches had no consistent impact on vegetation in the 3 years following blocking (Green et al., 2017). Even after six years, drainage blocking may not fully restore the water table of a peatland. One study found that after six years the behaviour of the water table of a drain-blocked peatland was intermediate between that of a drained and an intact peatland (Holden et al., 2011). In another study of a range of restoration techniques, the conclusion was that there was not enough evidence to give a timescale of recovery as no studies have been going for long enough (i.e., greater than 20 years) (Shepherd et al. 2013).

Evidence for changes in the colonisation of water within these habitats indicated that even newly created pools with low macrophyte cover may be able to sustain substantial populations of larger fauna via algal primary production, consumption of detritus, and microbial processing of humic substances and methane (Beadle 2015). A separate study found that there was relatively rapid recolonisation of benthic invertebrates in streams after 3-11 years of blocking (Ramchunder et al. 2011). A study on reedbed that had been created from arable land 40 years prior to study but restored from dry state 12 years prior to study found that 12 years of restoration was not sufficient to entirely restore invertebrate communities in terms of abundance, but that all species had been restored by this point (Hardman et al. 2010).

The landscape scale study of Wicken fen which includes both hydrological management (relevant to **EHAZ-129EM**), but also includes low intensity grazing, found that although differences in vegetation were achieved from 5 to 60 years of restoration, plant communities remain different from the target fenland community and it is estimated that hundreds of years would be required before target vegetation is reached (Stroh et al. 2012).

3.9.1.5 Spatial Issues

No information.

3.9.1.6 Displacement

It is unclear whether protection of peatland or rewetting will have displacement effects. It could influence stocking densities and move grazing elsewhere.

3.9.1.7 Maintenance and Longevity

Drainage blocks can generally be removed at a later date, which could reverse any changes.

3.9.1.8 Climate Adaptation or Mitigation

Rewetting encourages the growth of wetland species and contributes to carbon storage.

3.9.1.9 Climate Factors / Constraints

Hydrological management is affected by water tables in surrounding areas which are themselves affected by climate factors and water and land use.

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

Upland farmers have quite readily taken up drainage blocking incentives in the past. Many upland regions are quite unproductive, so economic barriers to uptake may be small (Alison et al. 2019).

3.9.1.11 Uptake

No information.

3.9.2 ECCM-033: Restore peatland vegetation

3.9.2.1 Causality

Evidence for success of restoring peatland vegetation is limited but generally positive (**amber**) and context dependent (**T**). (Conservation evidence action 1822). See above **ECCA-007EM** for improving hydrological conditions to restore peatland vegetation. Here we consider only methods to restore the biodiversity of peatland vegetation. Evidence for beneficial effects of introducing mixed vegetation fragments (including seeds, rhizomes, seedlings and spores of other species even if dominated by mosses onto the peatland surface) indicates that it is generally successful for sphagnum and some other species. Work by Hinde et al. (2010) investigating sphagnum introduction via beads or strands showed that colonisation of Sphagnum was largely dependent on weather at the time of spreading and just after. Hot dry weather at the time of spreading prohibited establishment. Bead application (stands of moss embedded in a gel) was assessed qualitatively to be much more successful than direct application of sphagnum strands, possibly due to enhanced desiccation resistance of the beads.

3.9.2.2 Co-Benefits and Trade-offs

No evidence.

3.9.2.3 Magnitude

Most studies indicate successful establishment of mosses or bryophyte species, particularly sphagnum and in one study, vascular plants.

3.9.2.4 Timescale

Studies indicate that effects of introduction of sphagnum may last for at least 6 growing seasons (Conservation evidence action 1822). Two years of monitoring showed some successful establishment of Sphagnum through the application of beads, but that longer-term trials are required to assess long term viability (Hinde et al. 2010).

3.9.2.5 Spatial Issues

No information.

3.9.2.6 Displacement

Removal of vegetation (seeds, brash) from areas where peatland is in good condition could be damaging, so care must be taken not to remove too much from any one area.

3.9.2.7 Maintenance and Longevity

See timescale. Suitable peatland conditions are required for longevity.

3.9.2.8 Climate Adaptation or Mitigation

Restoration of peatland vegetation contributes to carbon storage. Natural Flood Management.

3.9.2.9 Climate Factors / Constraints

Undrained wetlands are likely to remain climatically suitable for restored vegetation.

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

Costs of seeding vegetation (whether through propagating (e.g., sphagnum) or harvesting material from other areas of bog) to transport and spread at site need to be considered.

3.9.2.11 Uptake

No information.

3.9.3 ECCM-031; ETPW-155; ETPW-158 & ECCM-034: Actions for grazing on peatland

ECCM-031	Use controlled grazing (bogs and peatlands)
ETPW-155	Remove grazing from recovering peatland, susceptible habitats and sensitive vegetation
ETPW-158	Manage the dominance of graminoid or ericaceous species on bog by hydrological restoration, light summer grazing and cutting
(**ECCM-034**	Remove non-peat habitat vegetation (may not be by grazing)

These options relate to grazing on semi-natural habitats, primarily peatlands. They are being considered together because they all apply to the application of an appropriate grazing regime to peatland habitats. (**ECCM-034** may involve grazing or other management).

3.9.3.1 Causality

Impacts can be quite complex; hence the evidence is considered to be (**amber**) with limited evidence and context dependent (**T**). Controlled grazing, the removal of grazing, or using grazing to manage the dominance of graminoid or ericaceous species should have benefits to biodiversity[16]. Actions needs to be site specific, rather than generic, hence a results-based approach may be suitable. Controlled grazing suggests that the appropriate grazing regime will be applied to each site, so we have reviewed accordingly. Keenleyside et al. (2019) summarise available evidence on the conservation impact of grazing on peatland. They cite a review of evidence by Natural England on the conservation impact of moorland grazing and stocking rates in England (Martin et al., 2013), which found that: "moderate" and "variable" (spatially and temporally) levels of grazing are the most appropriate for delivery of many ecosystem services (including biodiversity). Martin et al. (2013) also found evidence that the habitat condition of previously low productivity blanket bog and montane habitats improved where stocking rates were reduced to annual averages of around 0.05 LU ha-1 yr-1 or less, often with off-wintering; and that similar stocking rates have allowed some recovery of previously suppressed montane plants in some of England's rarest and most fragile upland habitats (Martin et al., 2013). Silcock (2012) found that upland habitats such as dry heath, wet heath and blanket bog recovered because of reduced

[16] https://www.cwmidwal.cymru/en/managing-the-reserve/

grazing, by sheep in particular. Although reducing overall stocking levels from levels perceived to be excessive can result in habitat improvement, the issue is complex and dependent on both spatial and temporal variations in grazing pressure as well as on livestock species and breed. Martin et al. (2013) found evidence that a likely barrier to the achievement of biodiversity objectives, is variability in grazing pressure across a diverse grazing unit. The grazing patterns that result from sheep ranging behaviour and grazing preferences, management practices and topography are unlikely to match the conservation grazing requirements of different habitats and species. A reduction in sheep numbers, resulting either from conservation schemes or changes to farm enterprise structure, will not necessarily deliver grazing/conservation requirements fully. Silcock et al. (2012) found that under-grazing and loss of vegetation structure is now occurring in some areas, with adverse impacts for some species such as golden plover and other waders nesting in the uplands. They also found that a decline in hefting and shepherding has led to over-grazing and under-grazing on different parts of the same site. A challenge for conservation advisers and land managers is to better match livestock grazing patterns to the requirements of different habitats. Complete removal of grazing should only be applied in a targeted way and in the short-medium term. “It is likely that prolonged grazing exclusion could be detrimental in all but the very lowest productivity or most climatically suppressed habitats, as competitive species increase and gaps for colonisation by less competitive species are lost.” JNCC, reporting on its habitat surveillance and monitoring, has found that over-grazing of blanket bog results in loss of vegetation structure and of more palatable or vulnerable species (and their associated fauna), and the spread of rank, unpalatable plant species. In extreme cases, very heavy grazing and trampling can lead to exposure of bare peat and erosion. JNCC concludes that: “There is, therefore, a need for grazing to be undertaken at the right time and with the right intensity.”

ECCM-034 may be a necessary action to restore function and habitat quality to peatland habitats e.g., removing trees from bog (trees tend to dry-out the habitat and make it less suitable for biodiversity typical of bog habitats). Evidence for climate adaptation has been scored as (**amber**) because benefits from improving the condition of the bog need to be weighed against removal of trees.

3.9.3.2 Co-Benefits and Trade-offs

No evidence.

3.9.3.3 Magnitude

No information.

3.9.3.4 Timescale

In a study of fen restoration carried out through low intensity grazing, with no hydrological intervention, restoration took approximately 10 years (Large et al. 2007). The final state was restored fen with changes in NVC types towards wetter community types.

3.9.3.5 Spatial Issues

Grazing on the uplands is best managed with hefted stock - i.e., stock which has learnt where it is permitted to graze. Invisible fences and livestock collars may provide an alternative and are being investigated in various trials. However, as stated above, grazing patterns that result from sheep ranging behaviour and grazing preferences, management practices and topography are unlikely to match the conservation grazing requirements of different habitats and species.

3.9.3.6 Displacement

Reduced grazing in the uplands could lead to displacement effects with increased grazing elsewhere (Alison et al. 2019).

3.9.3.7 Maintenance and Longevity

No evidence.

3.9.3.8 Climate Adaptation or Mitigation

Maintaining an appropriate grazing regime can enable resilience to climate change (Natural England & RSPB 2019).

3.9.3.9 Climate Factors / Constraints

No evidence.

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

Economically speaking, upland farms struggle to profit without subsidies. While incentives to reduce stocking rates in upland areas might be readily taken up by some farmers, reducing stocking may run against some farmers' ideologies (Farming Connect, pers. comm.) or make businesses not viable. Anecdotally, hefting of sheep on common land may become increasingly difficult as sheep numbers diminish, which could result in abandonment of some upland areas (Alison et al. 2019).

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3.9.4 ECAR-041: Reduce managed burning on non-SAC/SPA designated sites and on shallow peat

This option can be cross referenced with options:
ETPW-143 Where burning takes place, ensure small burns on a long rotation to create a varied age structure in dwarf shrub, including retaining mature and degenerate phases; and
ETPW-144 Only burn in accordance with the heather and grass burning code.

3.9.4.1 Causality

Evidence is mixed **(amber)** for the impacts of burning on moorland and context dependent (**T**). Management by burning is not beneficial for some species, so an action to reduce burning should have some beneficial impacts on biodiversity **(**)**. Keenleyside et al. (2019) review impacts of burning and evidence from their review is summarised here.

Shaw et al., (1996) and Tucker, (2003) indicate that in appropriate areas and circumstances, carefully managed burning can play an important role in the maintenance of some open semi-natural upland habitats of high conservation importance so reducing burning could indicate some disbenefits. However, frequent burning and large fires, such as normally occur for agricultural management of moorlands, can result in declines in species richness. The burning of vegetation on peatlands is particularly damaging. Few studies have focused on habitat composition or biodiversity as a whole and instead monitor the impacts of burning on one species or group of species (Harper et al. 2018). More evidence is needed to determine the benefits/drawbacks of burning in comparison to other techniques (e.g., cutting, layering or grazing).

In the case of bird species, the creation of fresh palatable shoots of *Calluna vulgaris* for food and taller/older sections for nesting and shelter is highly beneficial to grouse (Glaves et al., 2013). Other species of bird, e.g., whinchat (*Saxicola rubetra*) and skylarks (*Alauda arvensis*), however, do not appear to benefit from prescribed burning as they are commonly associated with different sets of vegetation characteristics, which are not promoted by burn management (Pearce-Higgins and Grant, 2006). Tucker (2003) also suggests burning is detrimental for short-eared owls (*Asio flammeus*), hen harriers (***Circus cyaneus***) and merlin (***Falco columbarius***) if patches of older heath are not retained for nesting

purposes. Although (Baines *et al.* 2008), found that five bird species decreased following the discontinuation of moor management including rotational burning. Species diversity and richness could increase in habitats with a range of vegetation at different heights created by rotational burning practices (McFerran et al., 1995). Coulson (1988) suggested that under "good practice" burning regimes, terrestrial invertebrates are effective at recolonising areas as most are highly mobile. Relatively little is known about the impacts on whole invertebrate assemblages in upland habitats (moorland/peatland) making this a key area for future research. There is also a notable lack of studies addressing the impacts of burning on amphibians, reptiles or mammals within UK upland areas.
Reduction in burning may mean that large areas of old heather excluded from rotational burning pose a significant fire hazard (Davies et al. 2010).

3.9.4.2 Co-Benefits and Trade-offs

No evidence.

3.9.4.3 Spatial Issues

Most of the research and discussion on managed burning has been focused on areas such as the English Pennines which have been subject to over a century of grouse moor management, along with other human pressures such as air pollution, which have increased the cover of woody, fire-prone Calluna at the expense of moisture-retaining, C-accumulating sphagnum (Alison et al. 2019). In natural peatlands, the rapid vertical growth of sphagnum effectively limits the amount of woody biomass that is able to accumulate above the moss layer, making such systems intrinsically less fire prone. In addition, many areas of blanket bog that have been managed for grouse have also been subject to drying, either intentionally (via drains) or unintentionally (as the result of gully erosion), which increases the risk that wildfires will burn down into the organic soil. These highly modified systems may therefore require a level of management intervention and protection that other, less impacted blanket bogs do not.

3.9.4.4 Maintenance and Longevity

Evidence suggests that prescribed burning (proportion of area burned in a case study site) remains below recommended levels (Allen et al. 2016).

3.9.4.5 Climate Adaptation or Mitigation

Burning releases carbon but is likely to be more controllable than a wildfire burn which could end up releasing large volumes of carbon to the atmosphere.

3.9.4.6 Climate Factors / Constraints

Increased summer temperatures may result in higher likelihoods of wildfire burns. Climate change has already led to an increase in wildfire season length, wildfire frequency, and burned area.

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

Any reduction in burn frequency may involve a reduction in profitability (particularly of grouse moors) or of agricultural productivity.

3.9.4.8 Uptake

Evidence suggests that where prescribed burning remains below recommended levels (in terms of area burnt) it may reduce fuel load and promote biodiversity at the landscape scale (Allen et al. 2016). However, this is contentious and challenged by other authors, in some areas burning may be very intensive (Tucker, 2003).

3.9.4.9 Other Assessments

There is currently insufficient evidence to assess this action in terms of magnitude, timescale and displacement.

3.9.5 ETPW-153: Stabilise eroding peat through targeted restoration work

See also **ECCM-033** Restore peatland vegetation (3.9.2).

There is currently insufficient evidence to assess this action in terms of magnitude, timescale, spatial issues, displacement, maintenance and longevity, climate factors/constraints, or uptake.

3.9.5.1 Causality

There is limited evidence (**amber**) that practices which seek to stabilise eroding peat are likely to be beneficial. Stabilisation of eroding peat may result from; spreading heather brash, applying geojute (Hinde 2010), peat reprofiling (removing overhanging peat), re-vegetation (with heather, cotton grass or sphagnum (Shepherd, 2013; Caporn, 2007)) possibly with the addition of lime or fertiliser (Shepherd, 2013). In areas where natural succession is slow it may be assisted by planting with plug plants of bilberry, crowberry, hare-tail, common cotton grass and cloudberry. A study by Anderson et al. (2011) indicated that use of jute to stabilise the peat surface had no effect on vegetation cover. Cobbaert et al. 2004 found that fen plant cover did not significantly differ between mulched and unmulched plots.

3.9.5.2 Co-Benefits and Trade-offs

No evidence.

3.9.5.3 Climate Adaptation or Mitigation

Stabilised peat is less likely to erode resulting in loss of carbon and contributes to the protection of water quality (c.f. Dissolved Organic Carbon).

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

Actions may be costly.

3.10 Actions for Habitats with Specific Hydrological Characteristics: Habitat Creation, Management & Enhancement - Floodplains

3.10.1 ETPW-016C & ETPW-036EM: Actions for water meadows

ETPW-016C Create water meadows
ETPW-036EM Enhance, manage floodplain meadows

3.10.1.1 Causality

Evidence for this action has been assessed as (**green****) in terms of enhancing the condition of semi-natural habitats (Rothero et al. 2016). The evidence for the impact on rare or priority species is (**amber**), context dependent **(T**)**. This action involves restoring or creating traditional water meadows, also known as wet meadows, that have carefully controlled water levels. They provide valuable breeding habitats for wading birds and other biodiversity (Dicks et al. 2020). Creation/restoration actions may involve manipulating nutrient and water levels. If source material is not available, it may be necessary to carry out additional planting or sowing (Hölzel and Otte 2003).

There is evidence that bird numbers can increase after restoration of wet meadow (Lyons and Ausden 2005). Breeding wading bird numbers increased, with 15-20 pairs of northern lapwing (*Vanellus vanellus*) and 5-10 pairs of common redshank (*Tringa tetanus*) (Lyons and Ausden 2005). Other studies found that some species showed no change or decreases (Hellstrom and Berg 2001, Berg et al. 2002).

Management techniques successfully created/restored wet meadow plant communities in seventeen studies from France, Germany, the Netherlands, Poland, Switzerland and the UK (Dicks et al. 2020, Lyons and Ausden 2005). The techniques were <u>topsoil removal</u>, <u>introduction of target plant species</u>, <u>raising water levels</u>, <u>grazing</u>, <u>mowing</u> or a combination of <u>removing topsoil and introducing target plant species</u>, <u>plus livestock exclusion</u> (Berendse et al. 1992, Verhagen et al. 2001, McDonald 1993, Hedberg and Kotowski 2010). Three studies (one replicated controlled study and two reviews) from the Netherlands, Sweden, Germany and the UK found restoration of wet meadow plant communities had <u>reduced or limited success</u> (Klimkowska et al. 2007).

3.10.1.2 Co-Benefits and Trade-offs

No evidence.

3.10.1.3 Timescale

On Conservation evidence (Dicks et al. 2020) thirteen studies (five replicated and controlled of which two randomised) from France, Germany, the Netherlands, Switzerland and the UK monitored the effects of methods to restore or create wet meadow plant communities over a relatively short time period after restoration, and found some positive effects <u>within five years</u> (Berendse et al. 1992, McDonald et al. 1993, Jansen and Roelofs 1998). Three replicated studies from the Netherlands and Germany found restoration was not complete <u>five, nine or 20 years later</u> (Verhagen et al. 2001). A replicated controlled site comparison from Sweden found plant species richness <u>increased with time since restoration</u> (Lindborg & Eriksson 2004).

3.10.1.4 Other Assessments

There is currently insufficient evidence to assess this action in terms of magnitude, spatial issues, displacement, maintenance and longevity, climate adaptation/mitigation, climate factors/constraints, benefits and trade-offs to farmer/land manager or uptake.

3.11 Actions for Habitats with Specific Hydrological Characteristics: Habitat Restoration, Management & Enhancement - Coastal

3.11.1 ECCA-033EM: Manage/enhance coastal habitats to compensate for losses to climate change as part of a coastal management plan

Due to the breadth of these actions, it is not appropriate to consider magnitude, timescale, spatial issues, displacement, maintenance and longevity, climate adaptation or mitigation, climate factors/constraints, benefits and trade-offs to farmer/land manager or uptake.

3.11.1.1 Causality

Many of these are extremely broad actions covering all coastal areas. Evidence for different coastal actions is covered below. At the top level, **ECCA-033EM** has been assessed (**amber**) with limited evidence and context dependency (**L,T**) but with a likelihood of positive outcomes (*****, ****** or *******). The coding reflects the fact that this action is crosscutting across habitat types, i.e., for some habitats, e.g., salt marsh there is a good deal of evidence whilst for others e.g., shingle or coastal cliffs, evidence is more limited.

3.11.1.2 Co-Benefits and Trade-offs

[TOCB Report-3-6 Carbon **ECCA-033EM**] Coastal habitats are able to store significant amounts of carbon below ground, but the capacity for this varies substantially across the coastal habitat types in the UK. In most cases, evidence suggests that there is little potential for a change in management to significantly enhance rates of carbon sequestration in coastal habitats. However, there is a general lack of evidence for the impacts of coastal management on carbon sequestration. For reviews of the potential of coastal habitat creation to result in carbon sequestration see QEIA Report 3-6 on Carbon Sequestration.

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***

[TOCB Report-3-6 Carbon **ETPW-179EM**] There are no studies reporting the amount of carbon sequestered by shingle systems in the UK of which we are aware, constituting a significant knowledge gap of coastal system carbon (Beaumont et al., 2014; Parker et al., 2021). Vegetation that develops on shingle will sequester some carbon, but long term storage in this system is unlikely due to high levels of disturbance (Armstrong et al., 2020). Carbon stocks that subsequently enter the marine environment may be sequestered long term (Armstrong et al., 2020). More extensive shingle structures can develop permanent perennial vegetation, but rates of carbons sequestration are unknown.

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	**N**

*Duplicated evidence base: **ETPW-049** Control grazing on shingle*

3.11.1.3 Climate Factors / Constraints

The Natural Environment chapter of the UK Climate Change Risk Assessment Evidence Report (Brown, *et al* 2016) highlights that all coastal ecosystems are at high risk from climate change, due to the presence of flood defence and erosion protection structures, which prevent landwards rollback of the intertidal zone as a natural response to sea-level rise. Natural adaptive capacity is also limited by reduced sediment supply due to hard coastal defences (NE and RSPB, 2019).

3.11.2 ETPW-180EM & ECCM-046: Actions for saline & inter-tidal habitats

ETPW-180EM Enhance/ manage inter-tidal and saline habitats
ECCM-046 Use controlled grazing on intertidal, saline, salt marsh and coastal grassland habitats. This is duplicated **by EHAZ-101**: Control grazing on inter-tidal and saline habitats (below Mean High Water Springs to the sea)

3.11.2.1 Causality

The evidence for actions **ECCM-046** and **ETPW-180EM** is considered to be (**amber**), context dependent (**T**) and show some disbenefits (**D**) as well as some positive outcomes (******). There is evidence of successful restoration/maintenance and enhancement of saltmarsh habitats (e.g., Adnitt et al. 2007; Hudson et al. 2021). However, this is a complex area, involving the use of different management techniques which affect taxa differently. Evidence is often contradictory with reviews indicating that options aimed at enhancing or managing habitats have been unsuccessful in the past. Management techniques include grazing, vegetation planting, control and management of pollution events, management of freshwater input/drainage and management of access (Adnitt et al. 2007).

Grazing: Agri-environment schemes should provide a good method to improve habitat quality and prevent biodiversity loss by paying farmers to graze salt marsh habitats more sensitively. However, there is good evidence that previous AE options have not been successful (Mason et al. 2019, Malpas 2013).

A survey of 50% of salt marsh sites in England (Mason et al. 2019) found that although most saltmarshes suitable for grazing in England were grazed, the desired outcomes for conservation grazing were not being achieved. The survey report concluded that current/past agri-environment schemes are an ineffective delivery mechanism for conservation grazing on saltmarsh. There is also evidence that AE schemes did not successfully influence the decline of salt-marsh breeding birds between 1996 and 2011 (Malpas et al. 2013). Saltmarsh-breeding Redshank (***Tringa tetanus***) declines continue and are likely to be driven by a lack of suitable nesting habitat although other factors are likely to be involved e.g., disturbance, predation.

Increases in plant richness as a result of grazing may be offset by reductions in invertebrate richness and herbivorous invertebrate abundance. Ford et al. (2012) found that predatory, zoophagus and detritivorous Coleoptera, foliage running hunters, space web builders and larger bodied invertebrates were significantly more frequent/abundant on un-grazed marshes. However, predatory Hemiptera and Araneae (sheet weaver spiders) were significantly more abundant on grazed marshes. Ford et al. (2012) concluded that having a mixture of grazed and ungrazed saltmarsh habitat is important. Davidson et al. (2017) also found a trade-off with fish populations and grazing, with three fish studies responding negatively to grazing.

Mason et al. (2017) found that at a national level, the timing of grazing was particularly critical. In terms of timing winter grazing (November–March) prevents optimal sward regrowth and causes soil compaction, poaching and erosion, while grazing in spring causes bird nest losses to trampling, (e.g., Smart et al. 2005, Adnitt et al., 2007; Doody, 2008; Sharps et al., 2017). Grazing intensity is also important. Lower Redshank breeding densities were found on heavily grazed or ungrazed sites compared to those lightly or moderately grazed (Malpas et al. 2013). This is because the sward heterogeneity is not suitable for nesting and feeding (Malpas et al. 2013, Smart et al. 2005) and high stocking rates may increase nest losses from trampling (Smart et al. 2005) or if stock create pathways exposing nests, it may lead to nest predation. However, there is evidence from other sites that suggests that Redshanks can do well in ungrazed marsh, provided that the sward structure is favourable. Sharps et al. (2017) found that light grazing of saltmarshes increased the availability of nest sites for Redshank but reduced their quality. Sharps et al. (2017) found that even light conservation grazing at less than one cattle per hectare can reduce Redshank nest survival rates to near zero. Grazing during the main egg incubation period should always be avoided.

The production of flowers and fruits in salt marsh vegetation may be prevented by intense grazing (Kiehl et al. 1998). Low stocking densities have also been found to favour abundances of voles, pollinators and flowers (van Klink et al. 2016). Van Klink et al. (2016) found that high density horse grazing was detrimental to biodiversity, particularly vole density, and showed an interactive effect with stocking density for Asteraceae flower abundance. Horses have higher food intake and can graze closer to the ground than cattle.

Natterjacks (*Epidalea calamita*) require heavy grazing but in the upper marsh / transitions only (as mid / lower marsh will be too saline and at risk of tidal inundation) (Adnitt et al. 2007).

The optimal conservation grazing for salt marshes (as defined by Mason et al. 2019) would be by cattle on 'historically' grazed sites at "low/moderate" grazing intensity from April or June to October, with variable sward heights and retained standing vegetation crops in the resulting habitat (Adnitt et al., 2007; Doody, 2008; Lagendijk et al., 2017; Mandema et al., 2015; Sharps et al., 2017; van Klink et al., 2016). Although it should be noted that there is a high risk to breeding Redshank even at low stocking levels during the incubation period i.e., before July.

Rather than general prescriptions, at a site level, it might be better to specify grazing intensity, timing and stock type (Mason et al. 2019, Malpas et al. 2013) although there needs to be a balance between a highly detailed prescription and flexibility for implementation of options according to site conditions.

If a marsh already has sward heterogeneity and good habitat condition, then grazing may not be necessary as even light grazing could have negative impacts (Sharps et al. 2016). Higher structural and species diversity has been found in ungrazed marshes and the promotion of natural dynamics particularly for the lower and mid salt marshes may be a better recommendation than a grazing regime (Kiehl et al. 1998). A mosaic consisting of ungrazed and lightly grazed areas may be the most beneficial option at a site level (van Klink et al. 2016).

Beneficial use of dredged materials: In situations where local sediment transport has been modified, for example by the creation of hard sea defences, it may be possible to transport materials from local dredging operations to saltmarshes as a proxy for natural processes. This technique is most likely to be appropriate for allochthonous marshes with a high tidal range.

Planting: The majority of saltmarsh species do not form a persistent seed bank and in embanked areas the seed bank of former marshes is likely to decline rapidly (**see Saltmarsh creation 3.1.1.1).**

Sediment trapping: This approach is generally only suitable where approaches that work with natural processes are not viable" (Hudson et al. 2021).

Regulated tidal exchange: "RTEs are generally ineffective at restoring natural functioning (for example, uninhibited transfer of water, sediment and biota in and out of the intertidal), but can be effective at reducing peak water levels and can aid in the creation of intertidal habitats that would not necessarily be able to form through managed realignment" (Hudson et al. 2021).

3.11.2.2 Co-Benefits and Trade-offs

No assessed.

3.11.2.3 Climate Factors / Constraints

The Natural Environment chapter of the UK Climate Change Risk Assessment Evidence Report (Brown, *et al* 2016) highlights that all coastal ecosystems are at high risk from climate change, due to the presence of flood defence and erosion protection structures, which prevent landwards rollback of the intertidal zone as a natural response to sea-level rise. Natural adaptive capacity is also limited by reduced sediment supply due to hard coastal defences (NE and RSPB, 2019).

3.11.3 EHAZ-089; EHAZ-070EM & ECPW-083: Actions for sand dunes

EHAZ-089 Restore/ manage natural water flow in coastal habitats- sand dunes
EHAZ-070EM Enhance / maintain sand dune (physical manipulation)
ECPW-083 Control grazing on sand dunes

3.11.3.1 Causality

Hydrology is the dominant environmental gradient operating on dune slacks (Curreli et al. 2013) with maximum water level being the key variable. This action has been coded **amber, context dependent with (**) or (***)**. It is considered to be desirable but with some limitations on evidence. Precipitation and evapotranspiration balances (Ranwell, 1959), winter flooding, intensity of drought and persistence of waterlogging in the rooting zone during the growing season are important variables affecting vegetation, through impacts on germination and productivity (Ernst, 1990; Grootjans et al. 1998). Evidence shows that the timing and duration of these events can alter inter/intraspecific competition, thus changing community composition (Bossuyt et al. 2003, 2005).

Sensitivity of dune slack vegetation to hydrological change is detailed by Davy et al. (2010). Small shifts in water table can result in species change. An experimental study showed that a shift of 10 cm in the water-table regime resulted in competitive shifts in two dune slack species (Rhymes et al., 2018), while field survey evidence suggests 20 cm shifts in the four-year average water-table regime differentiate the main dune slack communities, and only 40 cm difference in regime separates the wettest from the driest dune-slack vegetation type (Curreli *et al*. 2013). This difference between communities in the long-term average regime has been confirmed using longer runs of data (9-year averages) in the Sand Dune Managers Handbook (Jones et al. 2021).

Groundwater fluctuations also control nutrient status: high water levels in slacks reduce the mineralisation of organic matter, maintaining low nitrogen and phosphorus levels (Lammerts and Grootjans, 1997). In contrast, management techniques which encourage natural sand mobility may guarantee natural renovation of young successional stages, allowing the formation of new blowouts or creation of new secondary dune slack habitat through natural dune dynamics (Davy et al., 2010; Stratford et al., 2007). Other management methods used to improve mobilisation in sand dune systems (such as sod cutting, removal of invasive scrub, etc.) may be useful to alleviate detrimental effects of climate change in the absence of natural mobility (Kooijman, 2004).

Management of adjacent land use could also have significant effects on dune hydrology. In a study by Jones et al. (2021) it was found that the overall influence of the adjacent coniferous forest on the hydrology may extend as far as ~500 m into the dunes. Model results suggest that the beneficial effect of current clearings and thinning extends at least 100m into the dunes, altering water tables by 20-30 cm, and extending to a lesser degree further into the dune system.

3.11.3.2 Co-Benefits and Trade-offs

No evidence.

3.11.3.3 Spatial Issues

Spatial context is important for actions on sand dunes. The site sits within a wider system and an understanding of onsite hydrology and surrounding land use is important.

3.11.3.4 Maintenance and Longevity

Dependent upon continuous review and manipulation of hydrology.

3.11.3.5 Climate Adaptation or Mitigation

Manipulation of hydrology should enable adaptation/mitigation to climate change. Habitat also vulnerable to altered coastal dynamics (Natural England and RSPB, 2019).

3.11.3.6 Climate Factors / Constraints

Climate change is predicted to cause major shifts in sand dune hydrological regimes, yet little is known about the tolerance of these communities to change, and their precise hydrological requirements are poorly quantified. The effects of climate change may be exacerbated by drainage or groundwater abstraction, and any form of water abstraction should be discouraged (Curreli et al. 2013, Bakker et al., 2006, Davy et al., 2010, Grootjans et al., 1996, Van Dijk and Grootjans, 1993).

3.11.4 EHAZ-070EM: Enhance / maintain sand dune (physical manipulation)

3.11.4.1 Causality

The action has been scored as (**amber**), with limited evidence (**L**) and context dependency (**D**) but positive benefits **(*****)**. Evidence exists on the factors operating on dune systems and potential actions to maintain and enhance them, including, e.g., the Dynamic Dunescapes Handbook which collates a lot of information on best practise for managing dune systems (Jones et al. 2021). However, the evidence can be regarded as amber because there is uncertainty, conflicting evidence, some of the proposed actions such as destabilisation and turf-stripping are relatively recent, and more research is required. More evidence on the interaction between drivers and management e.g., how climate change may influence dunes in the future and interacts with management would be desirable. The precise eco-hydrological requirements of dune slack communities are poorly, if at all, quantified in the UK (Curreli et al. 2013).

Although chronosequences have been used to understand dune development (Jones et al. 2010), there is still a lack of long-term studies. In addition, potentially novel conditions in the future (through climate change) may make it more difficult to predict future development using historical variation.

The two main options for reinstating natural dune dynamics are turf stripping/reprofiling on a large scale for sites inland and notching of the foredune for locations near the sea. Notching increases wind speeds and sand supply into certain parts of the site, while turf stripping removes the vegetation and soil that has stabilised parts of the dunes (Jones et al. 2021). Turf stripping can also increase sand supply to surrounding areas if location and conditions are favourable. Other actions to maintain or enhance sand dune habitats include mowing, restoration/alteration of the hydrological regime, removal of nutrients (?), grazing (see below) and vegetation management e.g., removal of non-natives or scrub clearance.

Dunes are part of a complex system and to determine appropriate management there needs to be a holistic approach (Jones et al. 2021), to understand how the dynamism of the system, hydrology, geomorphology, atmospheric pollution (which can lead to eutrophication and dune stabilisation), and climate interact with management to influence habitat condition across the system. It would be inappropriate to apply localised management without considering the system interactions. Research suggests, however, that, despite our capacity for large-scale engineered re-mobilisation (Curreli et al. 2013, Arens et al. 2004; Rhind et al. 2007), the system will respond within the bio-climatic envelope defined by current climate and future climate, which may speed up or slow down vegetation colonisation and growth accordingly and affect the outcome of any management activity.

Ideally all developmental stages of habitat should be present in a dune system e.g., bare sand, pioneer communities and early successional habitats, as well as more mature dunes (Jones et al. 2021). Management needs to be tailored to historical conditions, current conditions, and future conditions (anticipating that in the future climate change is likely to have large impacts). Many dune systems are now considered over-stable (Jones et al. 2010) and this is of conservation concern (Jones et al. 2004; Martinez et al. 2004; Rhind et al. 2007) due to a reduced area of early successional habitats which are important for many rare sand dune species such as the Natterjack toad (*Epidalea calamita*), Sand lizard (*Lacerta agilis*), obligate dune invertebrates such as the Sand Mining Bee (*Colletes cunicularis*) and plants such as Seaside Brookweed (*Samolus valerandi*). Over-stabilisation is also of concern because the lack of natural dune dynamics means that systems are not able to adjust to environmental change through the natural processes of dune mobility and erosion and scour down to the water table creating new habitat in the wake of migrating dunes.

Management of the pioneer disturbance communities rich in dune annuals requires maintenance of some bare sand and halting soil development through promoting small-scale disturbance e.g., by rabbits. Results investigating sand dune development show that soil development starts to accelerate once full vegetation cover is achieved, and that the transition from semi-fixed to fixed dune grassland depends on the degree of soil development (Jones et al. 2010).

In wet dunes the requirements of individual species of conservation interest such as the Red Data Book liverwort (*Petalophyllum ralfsii*), and the fen orchid (*Liparis loeselii*) suggest that management interventions such as mowing or turf stripping may be necessary to create the appropriate conditions in the absence of natural dynamics which might create new habitat.

3.11.4.2 Timescale

It takes at least 45 years to establish a fixed dune grassland community, which can then persist for at least a further 100 years, given favourable management (Jones et al. 2010). However, this is based on previous climatic conditions, and it may be more difficult to predict future conditions. Turf stripping, restabilisation, deep ploughing, topsoil inversion, scrapes may take 0-3 years for vegetation colonisation, with restoration of semi-fixed dune habitats taking typically 5-20 years; fixed dune habitats 40 years+ and 3-40 yrs for successionally young slack communities (Maskell et al. 2014).

3.11.4.3 Spatial Issues

Spatial context is important for actions on sand dunes. The site sits within a wider system and an understanding of the dynamism of the system, the sediment supply, whether the site is subject to accretion or erosion, potential impacts of sea level change are critical. In some dune systems restoration of natural function will be difficult because the dune is bounded by other non-sympathetic land uses e.g., urban development and this will affect management potential.

3.11.4.4 Climate Factors / Constraints

The Natural Environment chapter of the UK Climate Change Risk Assessment Evidence Report (Brown, *et al* 2016) highlights that all coastal ecosystems are at high risk from climate change, due to the presence of flood defence and erosion protection structures, which prevent landwards rollback of the intertidal zone as a natural response to sea-level rise. Natural adaptive capacity is also limited by reduced sediment supply due to hard coastal defences (NE and RSPB, 2019).

3.11.5 ECPW-083: Control grazing on sand dunes

We are unclear on what ‘control’ means in this context and have provided evidence on grazing to maintain biodiverse sand dune plant communities and associated fauna This action is duplicated by **EHAZ-067** Control grazing on permanent coastal grassland. The review below provides specific evidence in terms of magnitude, timescale, spatial issues and climate adaptation or mitigation.

3.11.5.1 Causality

There is good evidence (**green**) of the positive (*****, ******) effect of grazing on fixed and semifixed sand dune grasslands (Hewett 1985, Massey & Radley 1992, van Dijk 1992, Ford et al. 2012, Plassman 2010).

The main goals of grazing are to keep vegetation short and to control scrub growth. The shorter vegetation allows more light to reach the ground which helps rarer, less competitive species, to persist. Disturbance by grazers also creates small patches of bare soil which encourage germination from the seedbank, which helps to maintain plant diversity. Managed stock grazing (by domestic livestock) can keep the sward low, which encourages 'natural' grazers like rabbits. Rabbit grazing can keep the sward very short but tends to be patchy. This, together with fresh bare sand from their burrows helps create a mosaic of different habitats (Jones et al. 2021).

Without grazing, in all but the most highly dynamic systems, the plant community is likely to be dominated by highly competitive tall grasses or scrub (Janisová et al. 2011), plant diversity will be lower (Ford et al. 2012; Pykälä, 2003) with lower forb richness and cover. Grazing can enhance the abundance of sand dune species, and positive indicator species for habitat condition (Plassman et al. 2010). Invertebrate responses to grazing may be more mixed. Invertebrate abundance and diversity, particularly of large predatory spiders, carabids and staphylinids is often higher in ungrazed grasslands (Ford et al. 2012a; Morris 2000). It would be expected that pollinators would increase where forb richness was increased. Where grazing leads to a decline in 'tussocky vegetation' there may be negative consequences for invertebrates and reptiles (Newton et al. 2009). Natterjack toads are dependent on grazing of dune vegetation to maintain suitable terrestrial and aquatic habitat. In the absence of grazing, highly intensive management – cutting, mowing, pool creation / restoration will usually be required to maintain populations.

Grazing can either increase or decrease bird abundance and diversity dependent on feeding and nesting sward requirements (Vickery et al. 2001). In an experiment comparing different types of grazing (fully grazed' i.e., extensively grazed cattle, pony and rabbit grazed, 'rabbit grazed' and 'ungrazed' (i.e., abandoned), Ford et al. 2012 found that fully grazed grassland was significantly more species rich, particularly for forbs, than ungrazed grassland. Rotational grazing, where animals are moved at regular time intervals allowing vegetation time to 'recover', often has favourable effects on plant, bird and invertebrate diversity (Söderström et al. 2001; Wrage et al. 2011).

3.11.5.2 Co-Benefits and Trade-offs

No evidence.

3.11.5.3 Magnitude

In a study by Plassman et al. (2010) on grazing in dry dunes the total plant species richness increased by approximately 1.12 species per year, which was significant for all species groups except lichens, while in wet dune habitats, total species richness increased by 0.98 species per year, also representing a significant change for each species group. After the introduction of grazing management, in dry dune habitats the number of positive indicator species increased significantly by 0.65 species per year. In the dune slacks, the average number of positive indicator species increased significantly by 17.7% post-grazing by 0.13 species per year.

3.11.5.4 Timescale

It takes at least 45 years to establish a fixed dune grassland community, which can then persist for at least a further 100 years, given favourable management (Jones et al. 2010). However, potentially novel conditions in the future (through climate change) may make it more difficult to predict timescales for the future development of dunes using historical variation.

In the long-term study by Plassman et al. (2010), where grazing was introduced in the dry dunes, a steep initial increase in plant species diversity was followed by an apparent levelling off after 7 years.

3.11.5.5 Spatial Issues

Spatial context is important for these actions. Sand dunes sits within a wider system and an understanding of the dynamism of the system, the sediment supply, whether the site is subject to accretion or erosion, potential impacts of sea level change are critical. In some dune systems restoration of natural function will be difficult because the dune is bounded by other non-sympathetic land uses (e.g., urban development) and this will affect management potential.

3.11.5.6 Climate Adaptation or Mitigation

Climate change is predicted to cause major shifts in sand dune hydrological regimes, yet little is known about the tolerance of these communities to change, and their precise hydrological requirements are poorly quantified. The effects of climate change may be exacerbated by drainage or groundwater abstraction, and any form of water abstraction should be discouraged (Curreli et al. Bakker et al., 2006; Davy et al. 2010; Grootjans et al. 1996; Van Dijk and Grootjans, 1993).

3.11.6 ETPW-093: Enhance/ manage coastal cliffs and slopes

There is some evidence available for this action (Rees et al. 2015) although overall for maintaining biodiversity of semi-natural habitat it has been scored as amber L** and amber L* for rare and priority species. Habitat action plans for hard and soft cliffs have identified conservation actions. There will be a need to focus on soft cliffs linked to implementation of SMPs, the planning system and risk management. Many species need both cliff slopes and habitats on cliff tops, and it will be important to promote incentives that conserve both, (Rees et al. 2015).

3.11.6.1 Co-Benefits and Trade-offs

[TOCB Report-3-6 Carbon **ETPW-093**] Very little is known about carbons stocks and rates of sequestration in coastal cliff habitats (Beaumont et al., 2014; Gregg et al., 2021). As such, recommending specific managements or assessing the magnitude of their impact on carbon sequestration is not possible. Vegetation in coastal cliff habitats can vary from woodland, to scrub, to grassland and herb assemblages. A discussion of how managing these vegetation types can affect carbon stocks and sequestration can be found in QEIA Report 3-6 *Carbon Sequestration*. However, their applicability to coastal cliff habitats is not certain. Carbon stocks in cliffs are at a relatively high risk of erosion in parts of the UK (Moore & Davis, 2015), and as a result any carbon stocks in cliff systems may be at a relatively high risk of loss to marine systems in the future(Lim et al. 2015; Moore & Davis, 2015; Rhind, 2014).

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***

3.11.6.2 Climate Factors / Constraints

Climate change impacts are not well understood. There is some concern that increased rates of erosion resulting from changes in climate and sea level may be too rapid and impact on habitat quality (Rees et al. 2015).

3.11.7 ETPW-179EM & ETPW-049: Actions for shingle features

ETPW-179EM: Enhance/ manage shingle features
ETPW-049: Control grazing on shingle

Evidence relevant to these actions is extremely limited. The review below includes limited evidence in terms of displacement and climate factors/constraints.

3.11.7.1 Causality

The evidence for these options is (**amber**), the evidence is **limited** (not recent-the most recent evidence comes from a report for Natural England (Doody & Randall 2003)- and **context dependent**. Apart from the limitations of the evidence, what evidence there is suggests that actions may be ineffective and allowing natural processes to take place may be more successful/efficient. Shingle habitats are important for erosion prevention and for nature conservation and these roles can be conflicting (Doody & Randall 2003). The extent and nature of any restoration, post-excavation, depends on whether the excavation is above or below the water table. Where disturbance or excavation remain above the water table for most of the time then vegetation can, potentially, be restored on the shingle surface, although stable communities are likely to take a considerable time to revert to their natural state.

Unlike sand dunes, (and to some extent saltmarsh), plants play a limited role in stabilising the structure of shingle features. The creation of a natural beach profile is crucial to the establishment of shingle vegetation.
Recontoured beaches should be left to allow natural reshaping by winter storms before any attempt to plant vegetation is made (Walmsley & Davy 2001).

Natural regeneration of plant communities on shingle takes place over time and a variety of techniques have been tried to encourage the establishment of vegetation on dry degraded/disturbed surface shingle (although information is limited). These include use of sown seed for restoration (Walmsley & Davy 1997) and the use of container grown plants (Walmsley & Davy 2001). These do not appear to have been particularly effective, to date. Hence restoring vegetation to stable shingle may largely be a matter of 'leaving nature to take its course' (Doody & Randall 2003).

Restoring vegetation on shingle surfaces at or near the water table is more readily achievable, based on the limited examples from sites in the UK. The presence of moisture (a limiting factor in most undisturbed stable shingle) and the apparently more rapid build-up of humus help to create suitable conditions for plant establishment and growth.

Shingle structures may be grazed, but on most sites, vegetation is sparse, and grazing is not recommended.
Wherever possible from a nature conservation perspective, shingle structures should be left entirely alone and natural functions allowed to take their course (Randall & Doody 1995). Enabling space for natural hydrological and geomorphological conditions to operate by alleviating coastal squeeze may be the best strategy.

3.11.7.2 Displacement

Allowing natural landward movement of shingle features will, in some cases, affect other coastal habitats
such as saline lagoons, grazing marsh, fens and reedbeds, some of which will be designated sites. This is likely to be exacerbated in areas where there is coastal squeeze.

3.11.7.3 Climate Factors / Constraints

The Natural Environment chapter of the UK Climate Change Risk Assessment Evidence Report (Brown, et al 2016) highlights that all coastal ecosystems are at high risk from climate change, due to the presence of flood defence and erosion protection structures, which prevent landwards rollback of the intertidal zone as a natural response to sea-level rise. Natural adaptive capacity is also limited by reduced sediment supply due to hard coastal defences (NE and RSPB, 2019). Studies have established that there is a relationship between the rate of shingle (gravel) barrier retreat and the rate of sea level rise. A higher rate of sea level rise will be associated with faster landward movement. Movement of shingle features is likely to be accelerated by climate change resulting in sea level rise and increased storminess (Doody 2003).

3.12 Restoration Management & Enhancement of Semi-natural Habitats - Grassland

3.12.1 EBHE-226: Use rare breeds for conservation grazing

There is currently insufficient evidence to assess this action in terms of magnitude, timescale, spatial issues, displacement, maintenance and longevity, climate adaptation or mitigation, climate factors/constraints, benefits and trade-offs to farmer/land manager or uptake.

3.12.1.1 Causality

There is good evidence (**green**) of the positive (*******) effect of using rare breeds for conservation grazing. A Grazing Animals Project (GAP) Guide to Animal Welfare in Nature Conservation Grazing[17] provides practical advice to conservation managers and graziers and all keepers of livestock. Another GAP publication, the Breeds Profiles Handbook[18], gives brief descriptions of 55 breeds of livestock known, or anticipated, to be of value in conservation grazing. Many of these are rare or traditional breeds, as these have the characteristics that enable the stock to thrive on the nutritionally relatively poor forage afforded by many conservation sites. These characteristics are often identified as ‘hardiness’ and ‘thriftiness’ but are poorly defined except through the practical experience of conservation managers.

Conservation grazing cannot be filled by modern breeds or strains adapted to high-input, high-output systems. It is, therefore, a great opportunity for rare and traditional breeds, many of which developed in parallel with habitats now appreciated for their conservation value. This applies not only in the UK but also in other European countries. English Nature's Traditional Breeds Incentive for Sites of Special Scientific Interest, several grazing projects funded by the Heritage Lottery Fund and the Limestone Country Life Project, suggest that conservation grazing is no longer confined to nature reserves.

Conservation grazing can contribute to genetic conservation by:
• Enabling an increase in numbers and wider distribution of rare and traditional breeds.
• Allowing breeders to identify, and select, those individuals that fare best under relatively austere conditions.
• Providing an outlet, or providing additional grazing, for stock that could not otherwise be kept.
• Providing a market for good animals without reference to the showring.
• Providing a refuge for rare breeds from threats such as that posed by the National Scrapie Plan.
See (Small et al. 2008).

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17 [1] https://dnu7gk7p9afoo.cloudfront.net/Files/18.-a-guide-to-animal-welfare-in-nature-conservation-grazing.pdf

18 https://drive.google.com/file/d/13vQcYreLLqxXCJ5049K718lCdCJbJovz/view

3.12.1.2 Co-Benefits and Trade-offs breed

Not assessed.

3.13 Restoration Management & Enhancement of Semi-natural Habitats – Mountain, moor and heathland

3.13.1 ECPW-176EM, EBHE-216 & ETPW-142: Actions for enhancement and management

ECPW-176EM	Enhance or manage heathland (including heathland mosaics)
EBHE-216	Enhance or manage moorland (including common land), e.g., through appropriate traditional grazing techniques or Rewet moorland (including common land), e.g., through appropriate traditional grazing techniques
ETPW-142	Off-winter livestock or reduce winter grazing on upland and mountain heath
Also see	
ECCM-031	Use controlled grazing (bogs and peatlands)

There is currently insufficient evidence to assess this action in terms of magnitude, spatial issues, displacement, maintenance and longevity, climate adaptation or mitigation, climate factors/constraints or uptake.

3.13.1.1 Causality

Evidence is (**amber)** for enhancing or managing heathland (**ECPW- 176EM**) which is a very broad action under which a number of potential management actions may sit. There is conflicting evidence about the application of these differing management approaches (burning, controlled grazing, management of hydrological regime) and the ways in which they are applied (e.g. grazing intensity or timing) with variable impacts on different taxa[19] (Shaw et al. 1989). Available evidence from a meta- analysis suggests that grazing can result in an increase in the ratio of graminoids to ericoids on heathlands. However, there is very little evidence available on the relative impacts of burning, grazing and cutting on lowland heath[20]. Action **EBHE-216** is unclear (there are two differing versions), rewetting is scored as for **ECCA-007EM** which is primarily (**green**) or **(amber)** (for SSSI's and rare species) with (**), indicating that rewetting with low levels of grazing is likely to be beneficial for biodiversity. See Actions for habitat with specific hydrological characteristics. Peatland actions are reviewed in more detail in sections 3.8 and 3.9.

Action **ETPW-142** has been evaluated as (**green**) with (**). There is evidence that only winter grazing is inappropriate in upland areas where purple moor grass *Molinia caerulea* occurs because the absence of summer grazing allowed the deciduous *M. caerulea* to grow unhindered during the summer, whilst its main competitor ling heather *Calluna vulgaris* was subjected to winter grazing (Hulme et al. 2002). Setting appropriate stocking levels to maintain the condition of the vegetation must take into account site conditions (Hulme et al. 2002). In other studies where grazing was reduced or controlled there was an increased frequency of dwarf shrubs/*Calluna* and decreases in grass cover (Pakeman et al. 2003). Hence it is considered that reducing winter grazing is positive for biodiversity in upland areas including SSSI's (Martin et al. 2013).

[19] https://environmentalevidence.org/project/how-does-the-impact-of-grazing-on-heathland-compare-with-other-management-methods-systematic-review/

[20] https://environmentalevidence.org/project/how-does-the-impact-of-grazing-on-heathland-compare-with-other-management-methods-systematic-review/

3.13.1.2 Co-Benefits and Trade-offs

No evidence.

3.13.1.3 Timescale

Increase in the cover and height of dwarf shrub species can be seen in 4-5 years suggesting restoration is occurring but it will take longer to reach a fully restored state and may not be comparable to target community (Pakeman et al. 2003, Ross and Anderson 2011).

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

In a study for Countryside Council for Wales (Goodger and Toogood 2005) most farmers believed that sustainable heathland management carried out to achieve nature conservation objectives was only economically viable with financial support. The loss of livestock subsidies removed the incentive to keep stock on marginal, unproductive heathland.

3.13.2 ETPW-143 & ETPW-144: Burning

ETPW-143	Where burning takes place, ensure small burns on a long rotation to create a varied age structure in dwarf shrub, including retaining mature and degenerate phases
ETPW-144	Only burn in accordance with the heather and grass burning code.

Cross ref with Actions for Habitats with specific hydrological characteristics.

3.13.2.1 Causality

Evidence is mixed (**amber**) for the impacts of smaller burns, longer rotations and using the grass burning code on heathland, it is also context dependent (**T**) and may have disbenefits (**D**) as well as some positive (**) impacts. Evidence tends to refer to comparing burned with unburned rather than comparing different practises, logic suggests that smaller burns on a longer rotation will have lower impacts. It may also be the case that smaller burns could prevent larger wildfires that would have more detrimental impacts (Harper et al. 2018). It has been recommended that burn rotations should not be shorter than a 15–20 year reoccurrence on UK moorland, however, local conditions and vegetation types inevitably alter the appropriate return period. Carefully managed burning can play an important role in the maintenance of some open semi-natural upland habitats of high conservation importance and small burns in a long rotation could have beneficial effects on the age structure of ling heather *Calluna vulgaris*. However, a lengthening of the period between burns can lead to a slow decline of heather dominance as plants age. If burning is then resumed, individual plants of *C. vulgaris* may have aged beyond the stage at which regeneration readily takes place from stem bases, hence regeneration has to come from seed (Hobbs and Gimingham, 1984, 1987). Too frequent burning can result in the displacement of *C. vulgaris* by *Molinia caerulea* (Hulme et al. 2002, Hobbs and Gimingham 1987; Currall 1981, Harper et al. 2018). *C. vulgaris* is possibly the most commonly cited target species with regards to burn management. Some argue current burn practices reinforce the dominance of *C. vulgaris* creating habitats relatively low in species diversity (McVean and Ratcliffe 1962, Lindsay 2010) It has been suggested that site-appropriate burn rotational lengths to maintain the graminoid-Calluna balance and prevent loss of peat-forming Sphagnum sp. would be desirable (Harris et al. 2011).

The creation of fresh palatable shoots of *Calluna vulgaris* for food and taller/older sections for nesting and shelter is highly beneficial to grouse (Glaves et al. 2013). Other species of bird, e.g., whinchat (*Saxicola rubetra*) and skylarks (*Alauda arvensis*), however, do not appear to benefit from prescribed burning as they are commonly associated with different sets of vegetation characteristics, which are not promoted by burn management (Pearce-Higgins and Grant 2006). Species diversity and richness could increase in habitats with a range of vegetation at different heights created by rotational burning practices (McFerran et al. 1995). Coulson (1988) suggested that under “good practice” burning regimes, terrestrial invertebrates are effective at recolonising areas as most are highly mobile. Relatively little is known about the impacts on whole invertebrate assemblages in upland habitats (moorland/peatland) making this a key area for future research. There is also a notable lack of studies addressing the impacts of burning on amphibians, reptiles or mammals within UK upland areas.

3.13.2.2 Co-Benefits and Trade-offs

Burning releases nutrients from the plant material in the smoke and ash that may be washed off into the litter/substratum, but there is also potential for loss in runoff leaching or in the smoke (Shaw et al. 1989). Water supply catchments are at risk from water quality impacts of fire (Harper et al. 2018). Burning on peatlands reduces above-ground carbon stocks through the combustion of vegetation and has the potential to reduce the carbon storage in surface peats (Ward et al. 2007, Glaves et al. 2013). Loss of carbon from prescribed fires could be a necessary and beneficial reduction in fuel load, reducing the probability of a wildfire which would have a more detrimental effect on the carbon budget (Harper et al. 2018).

3.13.2.3 Other Assessments

There is currently insufficient evidence to assess this action in terms of magnitude, timescale, spatial issues, displacement, maintenance and longevity, climate adaptation or mitigation, climate factors/constraints, benefits and trade-offs to farmer/land manager or uptake.

3.14 Restoration Management & Enhancement of Semi-natural Habitats – Riparian Habitats

3.14.1 ECPW-291C & ECPW-219EM: Create and manage riparian habitats

ECPW-291C Create riparian habitats
ECPW-291EM Enhance or manage riparian habitats

3.14.1.1 Causality

The creation of riparian habitats (**ECPW-291C**) has been scored as (**amber T ***) for maintaining biodiversity under a changed climate, (**green *****) for enhancing the condition of semi-natural land and (green*) for presence of rare species and connectivity and (green**) for abundance of pollinators. Because of the potential to encourage non-native invasives, like Himalayan balsam (*Impatiens glandulifera*), it has been scored as (**amber DT***) for INNS. Restoration of habitats including woodland and forest along rivers is likely to be beneficial across a range of taxa[21]. Potential negative effects of riparian habitats on farmland such as invasive species are signalled by a (**red**) score for enhance condition of agricultural land. Enhancement or management of riparian habitats (**ECPW-291EM**) scores are similar for the condition of semi-natural land and rare species, however, connectivity, enhance condition of agricultural land and INNS are not scored as these habitats are already present and therefore unlikely to have the same impacts as their creation would.

3.14.1.2 Co-Benefits and Trade-offs

Riparian vegetation can strengthen the riverbank and prevent erosion and provide buffer strips for nutrient pollution.

[21] https://www.conservationevidence.com/actions/1416

3.14.1.3 Other Assessments

There is currently insufficient evidence to assess this action in terms of magnitude, timescale, spatial issues, displacement, maintenance and longevity, climate adaptation or mitigation, climate factors/constraints, benefits and trade-offs to farmer/land manager or uptake.

3.14.2 ETPW-006 & ETPW-067: Monitor and control damaging riparian species

ETPW-006 Monitor and control damaging aquatic animal species
ECPW-067 Monitor and control damaging aquatic plant species

3.14.2.1 Causality

These actions have been scored as (**amber L***) for biodiversity adaptation and maintain good condition of semi-natural habitat because of limited evidence. Scores for enhancing the condition of semi-natural habitat and SSSI's and rare species and the effect on invasive non-natives are all judged to be (**amber*****). Whilst these actions are very likely to have a positive impact on native biodiversity through reduced competition with aggressive invasive species there is little published scientific evidence for the benefits of monitoring and controlling aquatic species (Marbuah et al. 2014) who note that 'On the other hand, the literature on how to mitigate established species, by control or adaptation, is much less extensive. Studies evaluating causes for success or failure of policies against invasive in practice are in principle non-existing.'

In contrast, there are excellent guides to the effective removal of damaging species[22] outside of the scientific literature, but it seems that these efforts are seldom accompanied by monitoring.

3.14.2.2 Co-Benefits and Trade-offs

[ToCB Report-3-6 *Carbon* **ECPW-067**] Invasive aquatic plant species are capable of sequestering significant carbon. However, the fate of that carbon is unclear. *Egeria densa* in the US has been shown to contribute to carbon storage in sediments at significant rates (Drexler et al. 2021). However, the effects of the elevated respirations rates from the decomposition of aquatic plant material are well documented and high densities of invasive aquatic plants can significantly decrease levels of oxygen in the water, with large consequences for food chains (Ribaudo et al. 2018, Schultz and Dibble 2011). Elevated rates of plant respiration in invasive aquatic plant communities has been associated with high rates of CO, CH_4 and ammonia emissions in southern France (Ribaudo et al. 2018). The net effect on carbon balance was highly seasonally dependent, resulting in net sequestration in summer and net emission in winter (Ribaudo et al. 2018).

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

22
https://www.crew.ac.uk/sites/www.crew.ac.uk/files/sites/default/files/publication/CRW2016_05%20Final%20report_0.pdf
http://nora.nerc.ac.uk/id/eprint/509613/1/N509613CR.pdf

3.14.2.3 Other Assessments

There is currently insufficient evidence to assess this action in terms of magnitude, timescale, spatial issues, displacement, maintenance and longevity, climate adaptation or mitigation, climate factors/constraints, benefits and trade-offs to farmer/land manager or uptake.

3.15 Natural Regeneration – Rivers and water courses

3.15.1 ECCA-006, EHAZ-103, ECPW-066, EBHE-126, ECPW-068, ECPW-069 & ECPW-070

ECCA-006	Re-naturalise river catchments by, for example, reconnecting rivers with their floodplain, restoring and realigning rivers and restoring associated floodplain habitats
EHAZ-103	Reinstate more natural river function and form, including flow, depth and substrate form and processes
ECPW-066	Reinstate river meanders
EBHE-126	Manage realigned rivers to maintain natural flow
ECPW-068	Reinstate pool riffle sequence
ECPW-069	Re-naturalise bed levels
ECPW-070	Re-naturalise bank profiles (e.g., where over-deepened or straightened)

The consideration of beavers as agents of river restoration and management was outside the scope of this evidence review.

3.15.1.1 Causality

ECCA-006 has primarily been scored as (**green**) with (**** or *****) for the evidence of positive impacts from re-naturalising rivers catchments on species adaptation to climate change, connectivity and enhancing the condition of semi-natural and agricultural habitats. There is less evidence of the impacts on SSSI's rare species and pollinators, which are likely to be context dependent. Similarly for invasive non-native species, where there may also be potentially negative effects. All other actions (**EHAZ-103, ECPW-066, EBHE-126, ECPW-068, ECPW-069 and ECPW-070**) have been grouped for assessment. Scores for these options are similar or the same as those above and are predominantly (**green** and **** or *****). Connectivity of small feature habitats and increased abundance of pollinators are not scored, because actions are focused on the river rather than adjacent habitats. Evidence provided here applies across all options. Frequently the purpose and hence focus of evidence on re-naturalisation of river catchments is for flood risk management (e.g. Connelly, 2020). Burgess-Gamble et al. (2018) present the evidence base setting out the current state of the scientific evidence underpinning 'working with natural processes'. Although the focus of these projects is on flood risk, evidence on the wider benefits is summarised and much of what is presented here comes from this source. The effectiveness of measures is site-specific and depends on many factors, including the location and scale at which they are used. Other useful resources are included in footnotes[23,24,25]

River restoration: **EHAZ-103**, **ECPW-066**, **EBHE-126**, **ECCA-006**. Increased complexity of morphology following restoration induces a greater diversity of flow velocities, increases the range of physical habitat types, providing spawning sites, refuges and pools for a range of macrophytes, invertebrates, mammals, fish and vegetation (Gilvear et al. 2000, Arscott et al. 2005, Pederson et al. 2006, Addy et al. 2016).

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[23] River Restoration Manual https://www.therrc.co.uk/manual-river-restoration-techniques

[24] Floodplain Meadows Handbook: http://oro.open.ac.uk/60122/1/Floodplain%20meadows-new%20links-final-er.pdf

[25] Green Approaches to River Engineering https://eprints.hrwallingford.com/1250/1/Green_approaches_in_river_engineering.pdf

There are numerous examples of river restoration projects, including an annual river restoration prize for the best projects[26]. A site at Swindale Beck was developed to re-meander a historically straightened section of river[27]. The beck was heavily rock armoured on both sides with pronounced levees which meant that the straightened channel was cut-off from the surrounding floodplain; it was fast flowing and when water did overtop the levee it could not easily drain back into the channel. A study relating river flow to invertebrate ecological communities, including Swindale Beck as a sample site, found that highly modified flows, such as those observed within impounded systems, are likely to result in ecological communities different from those which might be expected under the natural flow regime (Hough et al. 2020). Dunbar et al. (2010) show that macroinvertebrate communities in rivers with high River Habitat Survey modification scores alter more in response to flow change than natural rivers. The River Cole restoration project was included in an EU LIFE funded urban and rural river restoration demonstration initiative between England and Denmark (Aberg et al. 2012). Evaluation showed a rapid positive change to physical habitat diversity, fish biomass and density returned to pre-restoration levels and there was an increase in plant species richness. Macroinvertebrates quickly recolonised, and one year following restoration species richness was only slightly below pre-restoration values and rarity of species was significantly lower compared with the prerestoration channel (Aberg et al. 2012, Addy et al. 2016).

Floodplain restoration: **ECCA-006**. Overall, there is a significant benefit for biodiversity of restoring floodplains. Projects, like the Low Stanger floodplain project, aim to restore lost habitat as well as providing a river floodplain restoration. Floodplains provide: a habitat for waders, wildfowl and fish; a food source for mammals; and maintain invertebrate, arthropod and macrophyte communities (Biggs et al. 2014, Arscott et al. 2005, Pederson et al. 2006). Lower turbidity allows more light for growth and visibility for feeding. However, a few species, such as snipe, do not benefit from floodplain restoration (Smart et al. 2008). Full river reconnection is likely to offer maximum benefits, providing a range of habitat wetland features and continuity for migration.

Restoration of floodplain habitats: Wetlands includes priority habitats such as grazing marsh, fen, reedbed and lowland raised bog which are valuable habitats for biodiversity. See section 3.2. Studies have found that 75% of restored wetlands are used by migrating birds (O'Neal et al. 2008). Managed wetlands are potentially most beneficial as a diverse range of habitats can be created and maintained (Bruland and Richardson 2005, Armitage et al. 2007).

3.15.1.2 Co-Benefits and Trade-offs

[TOCB Report-3-5D Systems **EBHE-126**] This is assumed to refer to re-naturalising river courses. This process promotes distinctive floodplain biodiversity (vegetation, but presumably associated fauna as well) and landscape heterogeneity (Jakubínský et al. 2021) and provides Natural Flood Management (NFM).

3.15.1.3 Timescale

Timescales for river restoration will vary depending on the flow regime, channel boundary conditions and sediment supply. The rate at which a river and floodplain become reconnected varies between different river types and the types of restoration undertaken. In some cases, the effect is immediate and in others the river needs time to adjust morphologically before it is able to attenuate peak flows.

[26] https://www.therrc.co.uk/uk-projects-map, https://www.therrc.co.uk/uk-river-prize
[27] https://restorerivers.eu/wiki/index.php?title=Case_study%3ASwindale_Beck_Restoration

3.15.1.4 Maintenance and Longevity

Restored rivers should be more self-sustaining and reduce the need for maintenance if restored to natural form within a natural surrounding environment. By accepting natural river processes and forms in restored rivers, maintenance costs can be reduced compared with channelised rivers that require regular maintenance. Restored floodplains and their wetlands do not have a finite lifespan. If restored appropriately they should be self-sustaining.

3.15.1.5 Climate adaptation or mitigation,

Maintaining and restoring natural river processes constitute the most ecologically effective climate change adaptation measures for river ecosystems (Kernan et al. 2012).

3.15.1.6 Other Assessments

There is currently insufficient evidence to assess this action in terms of timescale, spatial issues, displacement, climate factors/constraints, benefits and trade-offs to farmer/land manager or uptake.

3.15.2 ECCA-008: Create/enhance/maintain high flow storage reservoirs

This action was considered only for co-benefits and trade-offs.

3.15.2.1 Co-Benefits and Trade-offs

[TOCB Report-3-5C Semi-natural **EBHE-126**] There is (limited) evidence that artificial reservoirs can have beneficial effects on water bodies in adjacent landscapes (**amber L ****) as well as on increasing the abundance and diversity of insects, aquatic plants and species that feed on these, thereby increasing landscape resilience for species dependent on water (Deacon et al. 2018). The effectiveness of reservoirs in this role depends on many factors including their size, depth, vegetation cover, how they are managed, etc. as well as their context in relation to other water bodies. Please see Habitat creation 'ponds' for general evidence on the advantages of creating water bodies for biodiversity. Reservoir releases can be used to assist fish during low flows/high temperatures.

3.16 Natural Regeneration - Woodland

3.16.1 ETPW-266: Use woodland management (UKFS) for target priority woodland species (Co-Benefits and Trade-offs only)

3.16.1.1 Causality

Not scored. This is a high-level action and many of the more detailed actions below will fall under UKFS woodland management. The United Kingdom Forestry Standard (UKFS) is the reference standard for sustainable forest management in the UK and should be used as a basis for woodland management. Advice is available under UKFS for specific habitat requirements of priority species associated with woodland to help inform management options. This action is too generic to review.

3.16.2 EBHE-198: Restore/manage ancient woodland with native broadleaf species

3.16.2.1 Causality

This action has been scored primarily as (**green ****), it has crossover with action **ECCM-049** natural regeneration (see section 3.4) as restoration/management may not necessarily involve planting, if sufficient seed sources are available in the locality. Also see **ECCA-026** (section 3.1.5). Restoration of native woodland using native species has been a common recommendation and using locally adapted seed could increase woodland resilience, during the initial phase of establishment, the trees will be well suited to the conditions that they encounter.

However, the pressures faced by woodlands in the future from climate change and tree disease may not be met by existing native species. Whittet et al. 2015 state that 'there should be consideration of the capacity for newly established populations of trees to survive immediately and amidst increasingly variable environmental conditions'. They state that this may involve some consideration of other sources e.g. populations with genetic resistance to changing environmental conditions, such as drought. Care should be taken, and source assessed independently for each site and circumstance. Evidence is currently limited, and it is proposed that more information needs to be collected on the genetic variability of tree populations and how they respond to changing environmental conditions (Whittet et al. 2015).

3.16.2.2 Co-Benefits and Trade-offs

No evidence.

3.16.2.3 Displacement

There should be no displacement as a result of this action because only existing woodlands are considered, unless this means that coniferous woodland for timber will need to be planted elsewhere.

3.16.2.4 Climate Adaptation or Mitigation

This action could contribute towards climatic adaptation/mitigation by building resilience as a result of diversifying the forest stand (Natural England and RSPB 2019)[28].

3.16.2.5 Climate Factors / Constraints

Climate factors that could affect woodland include increases in extreme events, climatic stresses such as drought, high wind, increases in pests and diseases. There may be constraints on choice of species from susceptibility to environmental conditions and disease, but these factors can inform the choice of species.

3.16.2.6 Other Assessments

There is currently insufficient evidence to assess this action in terms of, magnitude, timescale, spatial issues, maintenance and longevity, benefits and trade-offs to farmer/land manager or uptake.

3.16.3 ETPW-125: Coppice and thin trees

3.16.3.1 Causality

The evidence is (**amber T*****), benefits for biodiversity are predominantly positive, although there may be negative effects on shade dependent specialists. Coppice produces a larger proportion of young growth stages which favour bird species depending on open ground or dense low shrub growth. Fuller et al. (2014) reviewed the evidence for coppice management effects on birds and found that the principal impact on bird habitats, supported by strong evidence, was the temporary provision of dense young woody vegetation years after cutting. Young vegetation in coppice woodland is denser than equivalent vegetation on replanted areas after clear-felling. The short rotation of coppice woodland also leads to a higher proportion of young growth stages than high forest with clear-fell. This dense low

[28] http://publications.naturalengland.org.uk/publication/5679197848862720

vegetation principally benefits warblers and other shrub-nesting species. The few years immediately after harvest also provide conditions suitable for some ground-nesting species. Normal silvicultural thinning does not tend to stimulate sufficiently large changes in the understorey to affect most birds.

Thinning can have positive effects for priority woodland mammals such as increasing the bat species richness and activity by creating suitable habitat for commoner bats (e.g., common pipistrelle) (Carr et al. 2020). Providing younger growth stages (regeneration and shrub) through removal of the overstorey has been shown to be of critical importance to dormice (Goodwin et al. 2018) and of significant use to pine marten in more fragmented habitats (Caryl et al. 2012b).

There are many plant species associated with ancient woodlands (Ancient woodland indicators) that are not shade dependent and require gaps and disturbances that could be provided through coppicing or thinning (Kimberley et al. 2013; Beauchamp et al. 2020). Although with more ‘natural’ non-intervention regimes overall understorey vascular plant species richness declines there are specialised shade-tolerant biota including plants, fungi and invertebrates associated with dead wood that have been linked to lack of disturbance and increased shade (e.g. Hambler and Speight 1995, Beauchamp et al. 2020) so management could have adverse effects on these species.

3.16.3.2 Co-Benefits and Trade-offs

No assessment.

3.16.3.3 Displacement

Not applicable because only existing woodlands are considered.

3.16.3.4 Climate Adaptation or Mitigation

If thinning reduces pressure on species and improves condition, there are likely to be benefits in building resilience to climate change (Natural England and RSPB 2019).

3.16.3.5 Other Assessments

There is currently insufficient evidence to assess this action in terms of, magnitude, timescale, spatial issues, maintenance and longevity, climate factors/constraints, benefits and trade-offs to farmer/land manager or uptake.

3.16.4 ETPW-124: Create/ enhance/ manage rides, edge habitats and open space (for biodiversity including for pollinators)

3.16.4.1 Causality

Evidence for this action has primarily been scored as (**amber TD*****) and for connectivity of small patches of habitat as (**green****). There is good evidence that the maintenance of woodland rides, open spaces and structural diversity is critical for pollinators, flora, all taxa of woodland edge specialists and those that need open space for part of their lifecycle (Beauchamp et al. 2020). Increasing woodland edges can have disbenefits for woodland interior specialists including some mammals, where the effects can be both beneficial and detrimental depending on their trophic level and whether they are open habitat or woodland species.

Plant diversity reduces with shade so much evidence indicates higher species diversity of trees, shrubs, and herbaceous plants in woodland gaps, along rides and woodland edges. The impact of better lit conditions around the edges of woodlands interacting with high adjacent land-use intensity can however lead to reduced abundance of typical forest species in favour of nitrophilous species

(Chabrerie et al. 2013). Not all of the plant species thought to be most characteristic of ancient woodlands are strictly shade-dependent either. Many are associated with better lit gaps and rides (Kimberley et al. 2013, Hermy et al. 1999, Peterken and Game 1984, Brown et al. 2015).

Guidance on woodland management for pollinators produced by DEFRA (DEFRA and Forestry Commission 2014), Buglife (Falk and Buglife 2019) and Butterfly Conservation (Clarke et al. 2011) emphasises the importance of maintaining woodland edges, rides and clearings. Good management of rides and clearings for pollinators will maximise the area receiving sunshine, prevent “wind tunnels”, introduce sown wildflowers, and introduce or encourage broadleaved native shrubs and trees.

3.16.4.2 Co-Benefits and Trade-offs

[TOCB Report-3-6 *Carbon* **ETPW-124**] Where managing habitats for biodiversity includes promoting species that require woody biomass, or preserving existing biomass, this could benefit carbon storage, above and below ground (see QEIA Report 3-6 *Carbon Sequestration)*. The removal of vegetation to create open spaces is associated with a reduction in carbon stocks, and the loss of future sequestration potential (see 3.11.9.2). The negative impact on carbon could be minimised by minimising soil disturbance and preserving carbon stocks in removed biomass as long-life products or using them to offset fossil fuel use (Matthews et al. in prep.). If actions are carried out on a relatively small scale, the magnitude of carbon loss may not be nationally significant.

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

3.16.4.3 Timescale

Some biodiversity benefits would be apparent in years 0-5, from providing areas of open habitat within the woodland (Keenleyside et al. 2019).

3.16.4.4 Displacement

No displacement because only existing woodlands are considered (except in productive woodlands where area for tree growth may be sacrificed).

3.16.4.5 Other Assessments

There is currently insufficient evidence to assess this action in terms of, magnitude, spatial issues, maintenance and longevity, climate factors/constraints, benefits and trade-offs to farmer/land manager or uptake.

3.16.5 EBHE-196: Planted Ancient Woodland (PAWS) restoration

3.16.5.1 Causality

There is good evidence (**green** or *****) that this action will be beneficial to biodiversity. The restoration of plantations on ancient woodland sites (PAWs) presents a good opportunity to increase biodiversity (Beauchamp et al. 2020, Pryor, Curtis and Peterken 2002, Thompson et al. 2003, Harmer and Thompson 2013[29]). Many ecological features remain on PAWs sites, and they can recover with

[29] https://www.forestresearch.gov.uk/documents/6948/FCPG021.pdf

restoration, even as the plantation reaches maturity. The approach to restoration is important, with gradual opening of the canopy and change essential to conservation and preventing further damage and biodiversity loss. This intervention largely involves the removal of non-native trees and encouragement of natural regeneration of native tree species, to provide a more varied age structure (Beauchamp et al. 2020). Atkinson et al. (2015) compared clear-felling and gradual thinning approaches to plantation restoration, both methods can be used for woodland ground flora species richness. However, if increasing invertebrate herbivore species richness is a concern, the gradual thinning approach is more appropriate.

Fuller et al. (2014) reported that the impact of plantation conversion on bird habitats is determined largely by the proportion of canopy removed. Where the plantation is removed completely by clear-felling, the results are similar, so a temporary increase in open and low shrub habitat potentially benefiting associated bird species. Where the plantations contain native trees, usually only the non-natives are removed, resulting in changes to bird habitats similar to thinning and potentially having similar effects.

3.16.5.2 Co-Benefits and Trade-offs

No assessment.

3.16.5.3 Climate Adaptation or Mitigation

If PAWS restoration improves condition, there are likely to be benefits in building resilience to climate change (Natural England and RSPB 2019).

3.16.5.4 Climate Factors / Constraints

Climate factors that could affect woodland include increases in extreme events, climatic stresses such as drought, high wind, increases in pests and diseases.

3.16.5.5 Other Assessments

There is currently insufficient evidence to assess this action in terms of, magnitude, timescale, spatial issues, displacement, maintenance and longevity, benefits and trade-offs to farmer/land manager or uptake.

3.16.6 ECCM-053: Manage deadwood (where appropriate, remove diseased deadwood, leave healthy deadwood to contribute to carbon storage)

3.16.6.1 Causality

There is considerable evidence (**green***) that deadwood in forest systems provides resources and habitat for biodiversity particularly saproxylic invertebrates (Beauchamp et al. 2020). Hodge and Peterken (1998) noted that 34% of scarce woodland invertebrate species (264 out of 771) require deadwood. Beetles (Coleoptera) constitute a large proportion of saproxylic invertebrate species in forests. Saproxylic invertebrate diversity is considered to be under threat throughout Europe, due to increased removal of deadwood from landscapes and shifts toward intensive commercial forestry (Davies et al. 2008b). Condition data from the National Forest Inventory indicated that 80% of British woodlands were unfavourable for deadwood volume in 2010-15, while in Wales 45% of surveyed sites had no qualifying deadwood present (Ditchburn et al. 2020 a,b). This suggests that existing woodlands need to be better managed for provision of deadwood if the promotion of saproxylic invertebrate diversity is considered a priority (Jonsell 2012).

A meta-analysis by Lassauce et al. (2011) examined the correlation between deadwood volume and saproxylic species richness, reporting a positive relationship. However, past management and the types of deadwood were important, and saproxylic richness may not respond linearly to deadwood volume. Sandström et al. (2019) carried out a systematic review on the effects of dead wood manipulation on abundance and diversity of saproxylic insects and other groups. Enrichment of deadwood through creation (i.e., using in situ trees as a source) and addition (i.e., using wood from external source) had positive effects on abundance and richness of saproxylic insects, including rare species. This study also found that burning benefited saproxylic abundance and richness more efficiently than creation or addition of deadwood, with similar effect sizes from approximately half the enrichment of deadwood volume (Sandström et al. 2019). Consequently, quantity is unlikely to be as important as qualitative aspects of deadwood stocks, such as structural diversity and presence of deadwood at various stages of decay.

Deadwood also acts as a habitat and resource for earthworms in forest systems, but it is not typically assessed in studies of earthworm diversity (Ashwood et al. 2019). Ashwood et al. (2019) recorded 7 earthworm species present in deadwood microhabitat of an oak-dominated broadleaf woodland.

Fuller et al. (2014) found little information from the UK on the effect of dead wood retention or provision on bird habitats. Removal of brash or fallen trees can remove nesting cover for some bird species and the creation of standing dead trees or snags has the potential to create suitable nest sites for hole-nesting birds, but many of these will also nest in holes in live trees and artificial boxes. Snags also provide food such as the larvae of bark beetles for some birds. Only rare species, such as Lesser Spotted Woodpecker (***Dendrocopos minor*)** and Willow Tit (***Poecile montanus*),** are highly likely to benefit from creating dead wood, but the specific evidence for such effects is sparse.

3.16.6.2 Co-Benefits and Trade-offs

No evidence.

3.16.6.3 Climate Adaptation or Mitigation

Maintenance of dead wood should improve the resilience of dependent species, restore soils' organic content and improve the capacity for moisture retention (Natural England & RSPB 2019).

3.16.6.4 Other Assessments

There is currently insufficient evidence to assess this action in terms of, magnitude, timescale, spatial issues, displacement, maintenance and longevity, climate factors/constraints, benefits and trade-offs to farmer/land manager or uptake.

3.16.7 ECPW-044EM: Manage or enhance targeted woodland

3.16.7.1 Causality

This action has been merged with action **ECCM-051C** (see section 3.1.6), note that evidence for this action denotes a (**red***) score for the potential positive impacts on invasive non-native species. Some of the actions described above are more specific management actions that could be applied to targeted woodland. The benefits of targeted woodland as opposed to non-targeted are described under action **ECPW-044C** (see section 3.1.6). Management will require mitigation of some of the effects of targeting i.e., it is not possible to target all species. If targeted means 'smaller' this may result in issues with habitat area and population sizes.

3.16.7.2 Maintenance and Longevity

Significant investment in improved woodland management lends itself to permanence, once the initial decision has been taken, but the long-term benefits depend on continuity of the habitat management system by successive land managers over many decades. Also, the felling licence system precludes most farm woodland removal, although it does not prevent neglect (Keenleyside et al. 2019).

3.16.7.3 Climate Adaptation or Mitigation

Improving the condition of woodland and reducing non-climatic pressures can build resilience to climate change (NE and RSPB 2019).

3.16.7.4 Other Assessments

There is currently insufficient evidence to assess this action in terms of, magnitude, timescale, spatial issues, displacement, climate factors/constraints, benefits and trade-offs to farmer/land manager or uptake.

3.16.8 EBHE-140EM: Enhance/manage Ghyll woodland

3.16.8.1 Causality

There is good evidence (**green*****) that Ghyll woodlands in good condition are of benefit to biodiversity[30] , native trees, ground flora (bluebells, wild garlic) with high diversity of cryptogrammic plants (Waite et al. 2010) and bird species requiring open woodland; including Redstarts (***Phoenicurus phoenicurus***), Pied Flycatchers (***Ficedula hypoleuca***) and Wood Warblers ***Phylloscopus sibilatrix***. Scrub can be important for species such as Black grouse (***Tetrao tetrix***), Nightjars (***Caprimulgus europaeus***) and Stonechats (***Saxicola rubicola***). Enhancement or management of Ghyll woodland can involve reduction in grazing pressure (by fencing) which also protects stock, allowing natural regeneration where possible with sufficient source material, additional planting may also be required preferably from seed sources from similar areas. Other management could involve leaving dead wood, avoiding fires, controlling invasive species.

3.16.8.2 Co-Benefits and Trade-offs

No evidence.

3.16.8.3 Timescale

Decadal restoration trajectories (Hughes et al. 2005).

3.16.8.4 Spatial Issues

There should be no changes to connectivity unless habitat condition substantially improves.

3.16.8.5 Climate Adaptation or Mitigation

Improving the condition of woodland and reducing non-climatic pressures can build resilience to climate change (Natural England and RSPB 2019). Woodland habitat provides microclimate and shelter for species and could protect headwater streams and springs. Optimisation of riparian tree cover helps to provide patchy light and shade. This, in turn, provides the best mosaic of biotopes, an ample supply of woody debris and leaf litter, and provides buffering against rising water temperatures, shading the water and lowering temperature on sunny days (NE and RSPB, 2019).

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[30] https://www.rspb.org.uk/our-work/conservation/conservation-and-sustainability/farming/advice/managing-habitats/clough-woodland/

3.16.8.6 Climate Factors / Constraints

Climatic constraints particularly for wet woodland include drying out of sites reliant on rainfall which could lead to a change in the dominant tree species and conversion to drier woodland habitat types. The composition of ground flora is also likely to change. There could be increases in pests and diseases e.g. *Phytophthora* on alder (*Alnus glutinosa*)(Natural England and RSPB 2019).

3.16.8.6.1 Other Assessments

There is currently insufficient evidence to assess this action in terms of, magnitude, displacement, maintenance and longevity, benefits and trade-offs to farmer/land manager or uptake.

3.16.9 ECPW-071EM: Enhance or manage floodplain woodland

3.16.9.1 Causality

There is some evidence that enhancing or managing floodplain woodland can be beneficial for biodiversity in the right context (amber T**). Floodplain woodland comprises all woodland lying within the fluvial floodplain that is subject to a regular or natural flooding regime (Cooper et al. 2021). Floodplain woodland can be biologically diverse, comprising wet woodlands of alder and willow and drier broadleaved woodlands (oak, ash, black poplar, grey poplar). Floodplain woodland in the UK is fragmented and so this action may be more focused on creation than restoration. Ecological restoration of floodplain forest would only enhance biodiversity if the natural geomorphological heterogeneity of floodplains is restored, and floodplain use is reformed (Brown et al. 1997) the floodplain is often disconnected from the river by artificial structures (Cooper et al. 2021) which would require intervention. Biodiversity is enhanced by fluctuating disturbance regimes. The type of woodland would obviously be important, large woodlands or plantations would be unsuitable and ideally a mosaic of grassland and woodland (woodland <30% cover) would be optimal for biodiversity (Brown et al. 1997).

3.16.9.2 Co-Benefits and Trade-offs

No evidence.

3.16.9.3 Climate Adaptation or Mitigation

More frequent extreme events could create opportunities for restoring or creating floodplain woodland as a flood, erosion and water quality management tool (NE and RSPB 2019). Floodplain woodland dissipates flood energy, reduces flood velocity and increases local water depths. This can delay flood peaks, reduce downstream flood peaks but increase upstream flooding due to the backing up of floodwater[31,32] Within wet woodland, the retention of in-stream woody debris can help to enhance flood alleviation.

3.16.9.4 and Other Assessments

There is currently insufficient evidence to assess this action in terms of timescale, magnitude, spatial issues, displacement, maintenance and longevity, climate factors/constraints, benefits and trade-offs to farmer/land manager or uptake.

3.16.10 ECCA-028 & ECCM-054: Continuous cover actions

ECCA-028 Transform (native and exotic) plantation woodland to continuous cover system of management

[31] https://www.therrc.co.uk/blog/webinar-woodlands-nfm
[32] https://www.therrc.co.uk/sites/default/files/projects/28_cary.pdf

ECCM-054 Diversify woodland / forest / plantation stand structure and species, including by the use of continuous cover systems of management

3.16.10.1 Causality

There is good evidence (**green*****) that Continuous Cover Forestry (CCF), where suitable, reduces many of the negatives associated with clear fell management, although it needs to be balanced against the potential impacts of increased management on recreation and wildlife. Fuller et al. (2014) included a specific study of upland conifer plantations with a Sitka Spruce (*Picea sitchensis*) component in Perthshire, Argyll, Borders and North Wales, quantifying differences in species richness and abundance of breeding birds under Continuous Cover Forestry (CCF), and large-scale clear-felling and restocking. Ranking the forest types in descending order of species richness gave: CCF with shrub understorey>CCF without shrubs>young pre-thicket clear-fell>mature clear-fell. Many 'forest birds' were most abundant, or recorded only, within CCF (e.g. Willow Tit (*Poecile montanus*), Wren (*Troglodytes troglodytes* ssp. t*roglodytes*), Wood Warbler (*Phylloscopus sibilatrix*), Blackcap (*Sylvia atricapilla*), Redstart (*Phoenicurus phoenicurus*) and Hawfinch (*Coccothraustes coccothraustes*). A small number of 'young-growth' species were most abundant in pre-thicket. The review of woodland management in the same report found that CCF tends to favour bird species associated with closed canopy woodland. These patterns support the value of CCF for biodiversity as an option for forestry, and that this value is greater than conventional forestry practice.

3.16.10.2 Co-Benefits and Trade-offs

No evidence.

3.16.10.3 Climate Adaptation or Mitigation

Encouraging continuous cover forestry rather than large-scale clear felling increases the structural heterogeneity and builds resilience to climate change. Continuous cover forestry approaches may be more wind-firm, maintain a more even carbon storage, and promote recruitment by maintaining higher humidity levels (Kirby *et al* 2009, Natural England and RSPB 2019).

3.16.10.4 Other Assessments

We were unable to evaluate evidence to assess this action in terms of timescale, magnitude, spatial issues, displacement, maintenance and longevity, climate factors/constraints, benefits and trade-offs to farmer/land manager or uptake.

3.16.11 EHAZ-138 & ECAR-042: Wildfires

EHAZ-138 Manage vegetation to reduce the risk of wildfire &
ECAR-042 Create/ maintain fire breaks to minimise spread of wildfires

There is currently insufficient evidence to assess this action in terms of magnitude, spatial issues, displacement, maintenance and longevity, benefits and trade-offs to farmer/land manager or uptake. Minimal evidence on other factors is included below.

3.16.11.1 Causality

Fires will happen and are likely to be more frequent with climate change and possibly from a reduction in prescribed burning (Davies et al. 2008). Vegetation can be managed to reduce the risks from wildfires. The type of management will depend upon the habitat type. This may involve the targeted application of prescribed fire for fuel management (e.g., removing build-up of old growth heather). It could also involve re-wetting in blanket bog to make sure that the vegetation is damp enough not to burn[33].

Fire breaks could be strips cleared of vegetation, strips of fire-resistant vegetation, embankments, empty ditches or water-filled ditches. To be effective a fire break is required to be at least 2.5 times flame height (expected flame length). This is normally 6 m to 10 m wide to be reliable under all conditions[34]. Larch (Larix spp.), for example, has been used in plantations to suppress ground cover and to create firebreaks (Parsons and Evans 1977), and increased proportions or belts of native broadleaf deciduous species will have a dampening effect on fire spread. Fire can be used to create fire breaks and control zones that can help to prevent whole landscapes being lost in a single wildfire. A small quantity of burning alongside tracks and other natural firebreaks is advisable as part of a wider fire protection strategy in areas otherwise set-aside from burning. Two studies evaluated the effects on butterflies and moths of mechanically removing mid-storey or ground vegetation to create fire breaks, species richness was the same or higher in areas where fire breaks were created (Bladon et al. 2022). These actions have been scored as (amberTD**) for evidence and impact on biodiversity in terms of limiting potential fire damage (also see ETPW-143). The actions should be targeted. The D is included because preventing fire in some communities could adversely impact the habitat through build-up of dead vegetation (increasing future fire risks). Fire assists regeneration encourages seed germination and prevents succession to scrubland in some habitats (Gazzard et al. 2016). There is some evidence that some bird species declined on a grouse moor after fire suppression (Baines et al. 2008, Williams et al. 2020). There is limited evidence for impacts on other indicators considered.

3.16.11.2 Co-Benefits and Trade-offs

No evidence.

3.16.11.3 Timescale

Short time scales.

3.16.11.4 Climate Adaptation or Mitigation

Wildfires will increase stress and reduce capacity to respond to climate change so reducing the risk of wildfires will improve adaptation, although this may involve some small-scale burns that will have some climate change impacts (Davies et al. 2008).

3.16.11.5 Climate Factors / Constraints

Climate change is likely to result in increased frequency and intensity of wildfires (Natural England and RSPB 2019).

3.17 Natural Regeneration - Woody features

3.17.1 ECCM-056: Manage veteran and ancient trees

3.17.1.1 Causality

It is rather unclear what exactly ‘manage’ veteran and ancient trees means – we have taken it to involve preserving these trees, potentially through a range of different mechanisms. This action has been scored as (**green*** or ****** or *******) across a range of ecosystem services. Mature forests and veteran tree species support higher levels of biodiversity than younger stands. Support may be needed to preserve mature and ‘over-mature’ trees to allow them to reach veteran status. Ecological succession from

[33] https://www.moorsforthefuture.org.uk/our-purpose/reducing-the-risk-of-wildfire
[34] https://cdn.forestresearch.gov.uk/2002/01/fctn3.pdf

mature trees near the end of their life to younger trees which also support the same habitat can also be supported by management. Buffers and fences could be added to protect the feature.

3.17.1.2 Co-Benefits and Trade-offs

No assessment.

3.17.1.3 Climate Adaptation or Mitigation

Manage veteran trees to reduce the crown-to-root ratio and improve protection for individual veteran trees (Natural England and RSPB 2019).

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

Historical and cultural landmarks as well as improving biodiversity.

3.17.1.5 Other Assessments

There is currently insufficient evidence to assess this action in terms of magnitude, timescales, spatial issues, displacement, maintenance and longevity, climate factors/ constraints or uptake. Minimal evidence on other factors is included below.

3.17.2 EBHE-203EM & ETPW-112: Actions for scrub

EBHE-203EM	Enhance / manage targeted scrub
ETPW-112	Manage scrub to maintain, restore and enhance grassland condition and associated species populations, recognising its inherent value in providing shelter/structure/food and nesting resource

3.17.2.1 Causality

Also see (**EBHE-203C**). In these actions scrub has been considered as a beneficial habitat and the focus is on maintaining scrub (but not letting it take over). These actions have primarily been scored as (**green****), although there may be disbenefits in terms of impacts (negative) on rare species and (positive) on invasive non-native species (**amber D****)**.** There is good evidence that enhancing and managing targeted scrub (**EBHE-203EM**) has positive effects on biodiversity (Mortimer et al. 2000, Day, Symes & Robertson 2006). Scrub can be valuable to many different taxa and is generally considered an important component of many habitats although, without effective management, it also has the potential to invade and spread and lead to successional development damaging early successional habitats. Therefore, whilst management and enhancement of scrub is generally positive for wider biodiversity, as always with any habitat change there will be winners and losers. Scrub is an important habitat for several breeding and wintering bird species and is used as a safe roost site and a source of invertebrates or berries as food. Many invertebrates feed on shrubs and many more on the associated lichens, algae and fungi of the bark and wood. Scrub also provides sources of food and shelter to mammals e.g. badger, deer, rabbits, foxes, dormice, bats.

Plant species which exist at the edges of areas of scrub and epiphytic species may also benefit from the effective management of scrub. Scrub may be detrimental to reptiles and amphibians e.g. sand lizards and great crested newts, (hence (**amber D****) where it covers large continuous areas, although in general a mosaic of scrub with variation in structure is likely to be beneficial.

3.17.2.2 Co-Benefits and Trade-offs

Not assessed.

3.17.2.3 Timescale

Management of scrub (in terms of destruction) can have immediate impacts. Managing for growth of scrub may take time.

3.17.2.4 Displacement

No displacement, management of an existing feature.

3.17.2.5 Maintenance and Longevity

Continued maintenance of scrub habitats is required to prevent succession to woodland – management can include grazing/browsing, burning and water table management.

3.17.2.6 Climate Adaptation or Mitigation

Management of scrub would be beneficial to the condition of the underlying habitat. Scrub contributes to C sequestration (in biomass, soils, and harvested forest products) and management should aid sequestration. It also contributes to adaptation or mitigation within the habitat by providing canopy cover that will influence temperatures and microclimate and provide shelter to shade tolerant species. In wetland habitats scrub encroachment can lead to drying out and lowering of the water table which has implications for climate change adaptation and management should be able to reverse this process.

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

As for woodland (3.1.5).

3.17.2.8 Uptake

Primarily as for woodland (3.1.5).

3.17.2.9 Other Assessments

There is currently insufficient evidence to assess this action in terms of magnitude, spatial issues or climate factors/ constraints. Available evidence on other factors is included below.

3.17.3 ECAR-033EM & ECPW-080EM: Shelter belts

ECAR-033EM	Enhance/ manage shelter belts (tree, woodland, scrub, and hedgerow) with appropriate species composition near sensitive habitats And
ECPW-080EM	Enhance, manage, wind breaks

3.17.3.1 Causality

The logic chain and limited available evidence indicate that the presence of shelterbelts is highly beneficial for biodiversity within landscapes (Prevedello et al. 2017). **ECAR-033EM** is a very broad action, not specifying what the enhancements/management involve, so it is assumed that this refers to maintenance of these features only – in respect of adjacent habitats. See evidence presented for the creation of these features (**ECAR-033C**, **ECPW-080C**) which is relevant here. There is limited evidence for the impact and management of those features on sensitive habitats, hence these actions have been coded as likely to be positive (**amber**, with limited evidence (**L**) and targeted (**T**)). Wind breaks are there for reasons other than enhancing biodiversity – hence it is likely that their management and those of shelterbelts **ECAR-033EM** will be focused on protecting crops.

3.17.3.2 Co-Benefits and Trade-offs

[TOCB Report-3-6 *Carbon* **ECAR-033EM**] Allowing woody features to increase in height and width can increase total carbon stock, and total carbon sequestered (see coppicing and thinning and hedgerow management). Rates of sequestration may then decrease. The removal of woody biomass can result in higher average sequestration rates, be lower average carbon stocks (see Report-3-6 *Carbon*). However, removed biomass carbon contribute to net sequestration, depending on the type of wood products produced and potential product substitutions (Matthews et al., in prep.).

There is some evidence that maintaining greater stand diversity can increase the resilience of stands to pressures, but evidence is lacking the scale of shelter belts. There is some evidence that species mixtures can support larger carbon stocks per hectare.

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	**L***

*Duplicated evidence base: Enhance/ manage trees and shrubs around point-source polluters [**ECPW-156EM**]; Enhance, manage, wind breaks [**ECPW-080EM**]*

3.17.3.3 Other Assessments

There is currently insufficient evidence to assess this action in terms of magnitude, timescale, spatial issues, displacement, maintenance and longevity, climate adaptation or mitigation, climate factors/constraints, benefits and trade-offs to farmer/land manager or uptake.

3.18 Natural Regeneration - Woodland

3.18.1 ECCM-049 & ECCA-027: Regeneration of woodland

ECCM-049 Create woodland by natural regeneration

ECCA-027 Encourage diversification of the stand and continuity of canopy cover through natural regeneration of native species in semi-natural woodland

3.18.1.1 Causality

This action has been assessed as primarily (**amber**) with limited evidence (**L**) and context dependency (**T**) but a likely positive benefit (***) for biodiversity. Evidence exists that creating woodland by facilitating natural regeneration can have benefits for biodiversity (Postnote 2021) although Burton et al. (2018) suggest that more evidence is required, and that this is an important area for future research. Rewilding approaches that attempt to restore natural processes often involve natural regeneration. Biodiversity should benefit from the diverse structure of natural tree growth, and locally adapted seed could increase woodland resilience. Success depends on the availability of seed sources including the proximity of existing woodland. Natural regeneration could result in natural processes taking decades to create woodland or in the dominance of a single early-arriving species like birch (Postnote 2021). Passive rewilding on abandoned farmland can result in closed-canopy woodland with woodland structural characteristics although species composition may not reflect that of adjoining ancient woodland (Broughton et al. 2021). Mixed effects on taxa and uncertainty have resulted in an **amber** score for the biodiversity aspects of this action.

3.18.1.2 Co-Benefits and Trade-offs

No evidence.

This action has not been assessed in terms of magnitude, timescale, spatial issues, displacement, maintenance and longevity or uptake.

3.18.1.3 Climate Adaptation or Mitigation

Management actions that promote regeneration will ultimately lead to maintenance of populations, continued canopy cover, and continuation of woodland (NE and RSPB 2019).

3.18.1.4 Climate Factors / Constraints

Increases in extreme events, increased pressures from pests and pathogens and changes in rainfall and temperature will constrain regeneration (NE and RSPB 2019).

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

Deer browsing (eating tree vegetation, particularly young stems) needs to be prevented which may involve costly fencing or shooting.

3.19 Maintenance & Restoration of Habitat Features in Parks & Gardens

The Historic England 'Register of Parks and Gardens of Special Historic Interest in England', established in 1983, currently identifies over 1,600 sites assessed to be of particular significance.

These two actions (with the same code) have been split rather than reviewed independently as other actions (below) review individual parkland features in Registered Parks and Garden:

EBHE-311 Enhance/ maintain parkland features in Registered Parks and Gardens
EBHE-311 Restore/ enhance / maintain parkland features in Registered Parks and Gardens

3.19.1 EBHE-307: Retain mature and veteran standing trees in Registered Parks and Gardens

3.19.1.1 Causality

This action has only been assessed for the categories ‘Biodiversity adaptation’, ‘Maintain good condition of semi-natural habitat’ and ‘Presence of Priority species’. For the latter two, the action has been assessed as (**green**) with a positive benefit (******) for biodiversity because of the following evidence. See also **ECCM-056** Manage veteran and ancient trees. Retention of mature and veteran trees is important for biodiversity (Isted 2004), they have been found to support higher levels of biodiversity than younger stands including saproxylic invertebrates, epiphytes, fungi, lichen, bird populations and bat roosts (Kirby 1998). Although parkland may be subject to more intensive management than other sites, studies have found that old park trees are, on average, as valuable for faunal diversity as trees in more natural sites (Jonsell 2012, Sverdrup-Thygeson et al. 2010). Additional management may be needed to preserve mature and ‘over-mature’ trees to allow them to reach veteran status (Perry 2013).

3.19.1.2 Co-Benefits and Trade-offs

[TOCB Report-3-6 **EBHE-307**/**EBHE-311** and others] For more a more thorough review of the importance of retaining trees, including ancient and veteran trees for carbon, see Report-3-6 Carbon, section on “Restoration, management and enhancement – Woody features” and notes on the action code **Carbon-01**.

A review of carbon storage in UK habitats found no studies that focused on the carbon balance of parkland, and thus the carbon stocks of mature and veteran trees in Registered Parks and Gardens is unknown (Gregg et al. 2021). However, old growth trees and woodland contain a large volume of carbon in biomass, despite small or no net carbon sequestration in old growth habitats. Estimates for Wales suggest that preventing the loss of existing woodland carbon stocks could constitute an effective emissions reduction of -121 t CO2eq ha-1 yr-1 to a time horizon of 2050 (Matthews 2020). However, the significant of this action will be largely dependent on the density of trees present to start with. Furthermore, although the per-hectare mitigation potential is high, current rates of permanent woodland area loss are relatively low (Brown 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	**LT*** **
	Below ground carbon sequestration	**LT***

3.19.1.3 Climate Adaptation or Mitigation

Retention of old trees is beneficial for climate adaptation, maintaining microclimates, shade, and habitat for many species.

3.19.1.4 Climate factors/constraints

Evidence indicates that beech dominated wood pasture in the south of England will be increasingly vulnerable to drought, particularly on freely-draining soils and soils subject to seasonal water-logging. More generally, drought and an increased frequency of storms pose a threat to veteran trees, which are a distinctive feature of much wood pasture and parkland (NE and RSPB, 2019). Greater survival of tree pests, such as grey squirrel and species of deer, resulting in increased browsing and grazing pressure and reduced regeneration. (Read et al 2009). Drought and fire risk lead to increased loss of mature and veteran trees and loss of associated saproxylic invertebrates, lichens and fungi (NE and RSPB).

3.19.1.5 Other Assessments

This action has not been assessed in terms of magnitude, timescale, spatial issues, displacement, maintenance and longevity because these are existing features within habitats. No evidence was available on , benefits and trade-offs to farmer/land manager or uptake.

3.19.2 EBHE-308: Re-plant trees in Registered Parks and Gardens

3.19.2.1 Causality

This action has only for been assessed for the categories ‘Biodiversity adaptation’, ‘Connectivity’ and ‘Enhance good condition of semi-natural habitat’. The action has been assessed as (**amber**) with a positive benefit (******) for biodiversity adaptation based on logic chain evidence and otherwise as (**amber**) and positive (*****) for other categories. There is limited evidence about the impacts of re-planting trees in registered parks and gardens. However, it is likely that regeneration and replacement of aging stock in parks and gardens may be necessary, with due consideration paid to the species used for planting for the retention of appropriate cultural and historic landscapes.

3.19.2.2 Co-Benefits and Trade-offs

[TOCB Report-3-6 **EBHE-308**/**EBHE-311** and others] For a review of the potential carbon benefits of planting woodland and trees see Report-3-6 Carbon, sections on ‘Habitat Creation – Woodland’ and on ‘Woody Features’.

There is no evidence specifically available to assess the potential of tree planting specifically in Registered Parks and Gardens. However, there is good evidence that planting trees can sequester significant carbon above ground, with woodland creation resulting in the largest benefits. There is also good evidence for carbon sequestration below ground long term, but the initial response of soil carbon stocks to tree planting can be relatively large losses, and the magnitude of the increase will depend on the starting condition of the soil.

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	**LTD****

3.19.2.3 Other Assessments

There is currently insufficient evidence to assess this action in terms of magnitude, timescale, spatial issues, displacement, maintenance and longevity, climate adaptation or mitigation, climate factors/constraints, benefits and trade-offs to farmer/land manager or uptake.

3.19.3 EBHE-309: Maintain standing/fallen deadwood in Registered Parks and Gardens

3.19.3.1 Causality

This action has been assessed for the categories 'Biodiversity adaptation', 'Enhance condition of semi-natural habitat', 'Maintain good condition of semi-natural habitat' and 'Presence of Priority species'. For biodiversity adaptation the action has been scored as (**amber L*****), there is limited evidence that leaving deadwood can enhance biodiversity under a changing climate, but the logic chain and existing evidence indicates that effects are likely to be positive (see **ECCM-053**). Available evidence, primarily for woodlands/forests rather than parkland indicates that leaving deadwood is highly beneficial (**green** ***) for biodiversity, particularly those taxa which rely entirely on deadwood (e.g., saproxylic invertebrates, see (Jonsell 2012). In more intensively managed parks dead wood may be commonly removed so including an action to retain it is likely to be beneficial.

3.19.3.2 Co-Benefits and Trade-offs

[TOCB Report-3-6 **EBHE-309**] For a review of the potential carbon benefits of maintaining deadwood, see the write-up of action **ECCM-053** in Report-3-6 Carbon. There is no evidence specifically available to assess the potential of maintaining deadwood specifically in Registered Parks and Gardens. However, there is good evidence that preserving standing and fallen deadwood where naturally occurring will positively affect carbon stocks. The removal of deadwood, particularly tree stumps, can lead to soil carbon loss via erosion.

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	**LTD***

3.19.3.3 Climate Adaptation or Mitigation

Maintenance of dead wood should improve the resilience of dependent species, restore soils' organic content and improve the capacity for moisture retention (Natural England and RSPB 2019).

3.19.3.4 Other Assessments

There is currently insufficient evidence to assess this action in terms of magnitude, timescale, spatial issues, displacement, maintenance and longevity, climate factors/constraints, benefits and trade-offs to farmer/land manager or uptake.

3.19.4 EBHE-310: Protect existing trees to prevent damage from livestock and wild animals in Registered Parks and Gardens

3.19.4.1 Causality

This action has only been assessed for the category 'Maintain good condition of semi-natural habitat' and has been scored as (**green**) with a likely positive impact (*****). Grazing animals are often a key feature of parkland management, evidence from pasture systems with trees indicates that protecting trees to reduce damage from animals is a beneficial action (Uytvanck et al. 2008).

3.19.4.2 Co-Benefits and Trade-offs

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***

[TOCB Report-3-6 **EBHE-310**] Compaction and root damage are deleterious to woody vegetation, and can lead to symptoms of drought and nutrient deficits if severe (Kozlowski 1999). Beyond this, there is a lack of evidence for how damage from livestock affects sequestration in mature trees. There is good evidence from Europe that deer browsing can reduce sequestration rates in mature stands, due to removal of biomass and resource reallocation by trees (Barrere et al. 2019, Côté et al. 2004).

3.19.4.3 Other Assessments

There is currently insufficient evidence to assess this action in terms of magnitude, timescale, spatial issues, displacement, maintenance and longevity, climate adaptation or mitigation, climate factors/constraints, benefits and trade-offs to farmer/land manager or uptake.

3.19.5 EBHE-090: Establish/ maintain a continuous grass sward in Registered Parks and Gardens

There was limited evidence for all actions under this management bundle. We have not reviewed this action apart from a section on trade-offs/co-benefits:

3.19.5.1 Co-Benefits and Trade-offs

[TOCB Report-3-6 Carbon **EBHE-090**] Introducing vegetation cover to bare ground will provide a small increase in carbon sequestration rates above and below ground (see QEIA Report 3-6 *Carbon Sequestration*). Maintaining vegetation cover can also increase soil carbon stocks by reducing rates of erosion, if ground was previously bare, potentially restoring soil compaction, and as a result of increased inputs from litter and exudates (see QEIA Report 3-6). Where the sward has high diversity or nitrogen fixing species are introduced, the productivity of the system may be further enhanced (also see QEIA Report 3-6).

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

3.20 Systems Action /Mixed Systems & Cross-habitat Action

3.20.1 ETPW-241: Manage a decline in soil nutrient levels for habitats / species that need low fertility

3.20.1.1 Co-benefits and Trade-offs

This action has been assessed as primarily (**amber**) with possible disbenefits (**D**) but a likely positive benefit (******) for biodiversity. The action refers to land which has previously been managed for agricultural production being stripped of nutrients to allow for a more 'semi-natural' assemblage of plants including a wider range of species tolerant of low nutrient conditions. Nutrient stripping can involve growing nutrient hungry crops of grass/cereals etc without fertiliser to reduce soil nutrients (over one or several seasons) or in more extreme cases inversion ploughing, or turf and topsoil stripping (see Magnificent Meadows[35]). Reduction of phosphorus loads in soils through cropping without fertiliser can take many years depending on starting status, whilst nitrogen loads can be reduced more quickly. Use of inversion ploughing, or turf/soil stripping may be deleterious to wildlife, such as grassland fungi and insects (NE TIN054)[36]. Nutrient stripping has proved effective for the restoration of a range of habitats including water meadows where topsoil removal has been shown to rapidly reduce nutrient levels and increase flood frequency (Holzel and Otte 2003). Dicks et al. (2020) covers 22 studies of flood meadow restoration, with topsoil removal, together with addition of target plant species which have been considered to be 'likely to be beneficial for biodiversity'. Other evidence on grassland restoration indicates that topsoil removal may assist the establishment of specific plant communities (Kaule and Krebs 1989). Evidence for heathland indicates that soil stripping may have limited effects on desired biodiversity (Walker et al. 2007).

The following actions were considered for a full assessment under this management bundle:

ECCA-035 Prepare and implement wildfire management plans
ETPW-117 Manage mosaics and natural transitions to other habitats
EBHE-219 Install/manage invisible fencing
ETPW-272 Control bracken and scrub by targeted grazing and trampling

3.20.2 ECCA-035: Prepare and implement wildfire management plans

3.20.2.1 Causality

This action is a plan rather than an action and has therefore been assessed primarily as context dependent (T) (green). 'Planned' or 'prescribed' wildfires have been shown to improve bird diversity in mountainous areas of France (Pons et al. 2003) and have mixed impacts on birds and bats globally (Conservation Evidence (Williams et al. 2020, Dicks et al. 2020). They have also been shown to have

[35] http://www.magnificentmeadows.org.uk/assets/pdfs/Soil_Nutrient_Stripping.pdf
[36] http://www.adlib.ac.uk/resources/000/264/842/TIN054.pdf

mixed effects on biodiversity in the UK uplands (Harper et al. 2018), which indicates the need for strategic planning. Plans should include timing, length and intensity of burn.

3.20.2.2 Co-benefits and Trade-offs

No evidence.

3.20.2.2.1 Other Assessments

There is currently insufficient evidence to assess this action in terms of magnitude, timescale, spatial issues, displacement, maintenance and longevity, climate adaptation or mitigation, climate factors/constraints, benefits and trade-offs to farmer/land manager or uptake.

3.20.3 EBHE-219: Install/ manage invisible fencing

3.20.3.1 Causality

This action has been scored on a logic chain basis as this is all relatively new technology and evidence collection is currently ongoing, hence evidence is lacking. The action has been scored as (**amber**) with limited evidence and context dependency (**LT**) and the likelihood of positive effects (*****). Invisible fencing is an innovation that allows the control of cattle movement without the need for physical barriers. Cows may be fitted with collars, in open areas with a Geographical Positioning System (GPS) which will signal when a cow approaches a boundary and provide a mild shock. Under trees due to the intermittent GPS signal, an alternative method is to bury an electric cable in the soil surface that emits a shortwave radio signal which is sensed by a transponder on a cattle collar. The transponder emits a noise as a cow approaches the boundary and, if she does not turn back, it provides a mild electric shock. Advantages are that habitats where traditional fences cannot be used (e.g. due to aesthetics) can be grazed in a management system to target management at certain areas and species, as required[11]. As grazing can be an effective tool for biodiversity management, the use of invisible fencing in areas where actual fencing is undesirable is likely to be positive.

3.20.3.2 Co-benefits and Trade-offs

No evidence.

3.20.3.3 Other Assessments

There is currently insufficient evidence to assess this action in terms of magnitude, timescale, spatial issues, displacement, maintenance and longevity, climate adaptation or mitigation, climate factors/constraints, benefits and trade-offs to farmer/land manager or uptake.

3.20.4 ETPW-272: Control bracken and scrub by targeted grazing and trampling

3.20.4.1 Causality

The action has been scored as (**amber**) with limited evidence and context dependency (**LT**) and the likelihood of positive effects (*****). There is limited evidence (Stewart 2005) for controlling bracken and shrubs solely through targeted grazing and trampling, hence the assessment is also based on the logic chain. Evidence exists for the effects of chemical treatment (Asulam) and cutting, with targeted grazing and trampling used as follow-up treatments to chemical treatment (Pakeman et al. 2002). Targeted grazing and trampling may also be used in association with mechanical treatments such as ploughing and cutting (Argenti 2012).

Evidence shows that grazing is unlikely to result in eradication, but reduction of bracken could be achieved (Pakeman et al 2002). A change in grazing systems from heavier to lighter animals, with a lower trampling effect, and hence reduced damage to developing fronds reduced the negative impact on *Pteridium* performance (Pakeman and Marrs 1992). Pakeman et al. (1997) showed that targeted grazing can damage bracken buds and developing fronds which are close to the surface or recently emerged, with cattle more effective than sheep. Grazers also disturb and break up the litter (encouraging frost penetration to the rhizomes) thereby preventing bracken regeneration.

A recent meta-analysis of the effect of targeted grazing found that it significantly reduced undesirable plants and significantly increased plant species richness. However, further research is needed to differentiate temporary defoliation from actual plant mortality and to address longer term outcomes following grazing cessation (Marchetto 2021).

Other targeted grazing literature focuses on the importance of grazing with the proper stocking density or intensity, with the right frequency, and at the right time of year (James et al. 2017). High stocking densities for shorter amounts of time are generally recommended to increase consumption of targeted plants (Bailey et al. 2019). Findings provide support for the use of targeted grazing as a vegetation management
tool for ecological restoration but with requirements for further research.

As bracken at low levels does have some benefits for wildlife, diversifying bracken structure rather than eradication is likely to be the best option in some situations. As part of a habitat mosaic, bracken can be important for invertebrates. Schlegel (2021) found that ongoing rotational sheep grazing system was the most appropriate approach to reducing bracken cover for the preservation of specialised dry and semi-dry grassland animal target species, including Red List Orthoptera. Bracken can be beneficial to fritillary butterfly's, small mammals, some plant species and birds such as Whinchat, Tree Pipit, Yellowhammer and Nightjar (NatureScot[37]). In summary, limited available evidence and the logic chain supports the idea of using grazing and trampling to control bracken but more research evidence is required to assess long term implications and context where it may be most successful.

3.20.4.2 Co-benefits and Trade-offs

No evidence.

3.20.4.3 Other Assessments

There is currently insufficient evidence to assess this action in terms of magnitude, timescale, spatial issues, displacement, maintenance and longevity, climate adaptation or mitigation, climate factors/constraints, benefits and trade-offs to farmer/land manager or uptake.

3.21 Specific Wildlife Targeted Actions

3.21.1 ECAR-034 & ECAR-036: Slurry & manure

ECAR-034 Locate new slurry storage away from sensitive habitats and
ECAR-036 Avoid spreading of organic manures close to protected area sensitive to ammonia/sensitive habit

3.21.1.1 Co-benefits and Trade-offs

Actions (**ECAR-034** and **ECAR-036**) have been considered jointly. These actions have been assessed as (**amber),** with limited information and context dependency **(LT)** but with a likelihood of positive impacts (******). A very recent report by Carnell et al. (2021) concludes that mitigation of intensive local "hot spot"

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37 https://www.nature.scot/sites/default/files/Publication%202008%20-%20Bracken%20Control%20-%20A%20Guide%20to%20Best%20Practice.pdf

point sources such as slurry stores by up to 80% (depending on the system in use) can reduce elevated atmospheric concentrations at nearby designated sites considerably. Therefore, if slurry covers were prioritised close to designated sites, i.e. using a spatially targeted approach, this could make a considerable difference to those sites. Subsequent spreading techniques will further affect likely volatilisation and must also be considered with respect to potential influences on adjacent sensitive habitats. There are several papers which highlight the effects of N deposition on sensitive sites and the damage it causes to plant species and communities, for example, Payne et al. (2013), but none that show the direct effects of slurry store locations or proximal spreading of manures and slurries on adjacent biodiversity.

3.21.2 ECCA-034: Create, enhance, manage natural refugia

3.21.2.1 Causality

The action **ECCA-034** has been scored on a logic-chain basis because it is such a broad action, encompassing a huge range of possible actions. In all cases scores are (**amber**) because we have used logic chain evidence often with limited information and context dependency (**LT**) but with a likelihood of positive impacts (******).
Natural refugia will differ according to what species are taking refuge and may range from creating ponds to hedge planting, enhancement of areas of semi-natural grassland or woodland creation, etc. Many of the actions assessed above already will constitute the provision of refugia for species.

Due to the breadth of this action, it is not appropriate to attempt to consider magnitude, timescale, spatial issues, displacement, maintenance and longevity, climate adaptation or mitigation, climate factors/constraints, benefits and trade-offs to farmer/land manager or uptake in any detail.

3.21.2.2 Co-benefits and Trade-offs

No evidence.

4 KEY ACTION GAPS

These are significant gaps relating to semi-natural habitats. Due to time limitations, here we highlight a few key areas where we believe there is a need for actions under the agri-environment schemes to ensure the effective management of semi-natural habitats.

- Retention of high-quality biodiverse habitats – Are farmers going to be encouraged to maintain high quality semi-natural habitats – which may not require any actions (other than what they do already)? There is potential risk that such habitats would be lost if they were not adequately valued by the system, especially under post-EU membership changes meaning that farmers will not be in receipt of basic payments.
- There are some habitats for which there are no actions.
- There are no actions for farming systems which are (at least in part) specifically aimed at enhancing biodiversity. These include organic farming for which there is a lot of evidence for biodiversity benefits at the system level, but potentially other certified farming systems like pasture fed livestock for which such evidence is emerging. Systems which minimise inputs are very beneficial for biodiversity.
- Hedgerows – there are no actions specifically relevant to hedgerow management/enhancement. These are and have consistently been the most significantly funded (as in the most £ paid out) actions by landowners. They are also THE key landscape feature in terms of enhancing biodiversity in otherwise impoverished landscapes. Whilst just paying a farmer for having a hedge and letting it deteriorate into a line of trees and eventually being lost altogether isn't advised, options for keeping hedges within a management cycle ARE advised. This should include rejuvenating by regular trimming and periodic laying/coppicing, gapping up where hedge plants are missing and enabling larger hedgerow trees to become established within features or planting hedgerow trees specifically. Actions may also include double fencing to protect hedges from being damaged by sheltering livestock.
- Stone walls – there are no actions for maintenance/enhancement of stone walls. These are important features for landscape diversity as well as for many aspects of biodiversity (Ruas et al. 2022).

5 EVIDENCE GAPS

This review highlights numerous evidence gaps (i.e., actions scored with either red or amber (with or without an 'L') in the table and discussed in the review above. Often there is general evidence from the literature on semi-natural habitats and associated biodiversity relating to management approaches, but evidence relating to monitoring the actual effects of implementing changes is far scarcer – hence many actions are scored amber. Evidence for the impacts of many of these actions on rare species is often limited. This is likely to be at least in part due to the complexity inherent in natural (and agricultural systems) and issues such as multiple drivers of change, time lags, differences between taxa in responses. These findings emphasise the importance of improved consistent long-term monitoring of wider ecosystem change as well as monitoring of specific agri-environment scheme actions.

There are some specific evidence gaps that can be identified; for instance, for coastal habitats there is limited evidence for some actions e.g. creating sand dunes and shingle, the physical manipulation of sand dunes. Some of the potential activities such as destabilisation and turf-stripping are relatively recent and this is a developing area of activity and research, more evidence should be available from projects such as Dynamic Dunescapes, if monitoring is successful. It is particularly important to understand coastal systems as focusing on individual coastal habitats will not be sufficient to deal with

challenges such as climate change. The proposed action for coastal management plans would be beneficial to address this.
Woodland creation actions have been coded as limited evidence, this is because there are many different factors to consider, such as how woodland creation influences all taxa, not just woodland specialists, which habitats are being replaced, indirect effects of new woodland on predators and invasive species. There is less evidence available for some specific woodland habitats e.g. Ghyll woodland and floodplain woodland. It has been identified that there is limited evidence on natural regeneration of woodland. This is important with the emphasis on woodland expansion usually through tree planting. This also links to landscape interventions such as rewilding.

There has been substantial research in peatlands and there is good evidence for many of the proposed actions. Some proposed actions have been developed more recently e.g. restoring peatland vegetation through sphagnum seeding, and stabilising eroding peat and although there may be a lot of experience in utilising these techniques there may not be peer reviewed evidence.

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