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Sustainable Farming Scheme Evidence Review Technical Annex

ERAMMP Report-25: Annex-4B: Building Ecosystem Resilience in Improved Farmland

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Approved by Simon Bilsborough

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Abbreviations Used in this Report

BBS	[BTO/JNCC/RSPB] Breeding Bird Survey
BGS	British Geological Survey
BTO	British Trust for Ornithology
CAP	Common Agricultural Policy
EAS	Environmentally Sensitive Areas
eftec	Economics for the Environment Consultancy
ERAMMP	Environment and Rural Affairs Monitoring & Modelling Programme
GHG	Greenhouse Gas
GMEP	Glastir Modelling and Evaluation Programme
HLS	Higher-Level Stewardship
IEEP	Institute for European Environmental Policy
N	Nitrogen
N2O	Nitrous oxide
PPP	Plant Protection Products
SFS	Sustainable Farm Scheme
UKCEH	UK Centre for Ecology & Hydrology
WBSM	Wild bird seed mix

Contents

1 Introduction

1.1 Review brief

This section of the evidence review complements ERAMMP Report 4: *SFS Evidence Review Annex 4 Building Ecosystem Resilience*, which had the remit to focus on management interventions that will deliver broadscale improvements in the biodiversity related features of semi-natural farmland and woodland habitat types - but specifically excluded improved farmland[1]. The Welsh Government now requires ERAMMP to undertake an evidence review based on the same brief as the earlier review, but this time to focus solely on improved land.

The requirement is to identify interventions and review the evidence for their effectiveness in building ecosystem resilience, which in this context refers to ecological resilience – defined in the State of Natural Resources Report (SoNaRR) (Natural Resources Wales 2016) as the capacity of ecosystems to deal with environmental disturbances, either by resisting them, recovering from them, or adapting to them, whilst retaining their ability to deliver services and benefits now and in the future. Ecological resilience is one aspect of supporting the change to more sustainable management of rural land in Wales that will be delivered through the Sustainable Farming Scheme.

1.2 Interpretation of the review brief

1.2.1 Improved land in Wales

Improved land in some parts of the UK can be characterised as enclosed farmland where the infield land has been agriculturally improved, is used to grow annual crops or grass leys and is maintained by regular management, which may include application of fertilisers (organic and inorganic), plant protection products (PPP), herbicides and intensive grazing or mowing; while semi-natural habitats may be found in field boundaries and in separate enclosures (e.g. farm woodlands, wetlands).

However, in Wales, the distinction between agriculturally improved land and semi-natural habitats can be less clear cut. In Wales there are areas of permanent grassland which can consist of continuously varying mosaics of improved or semi-improved land. For example, remaining low-lying wet grassland areas that are unresponsive to cost-effective agricultural improvement can be found embedded within an undulating, largely improved setting or adjacent to a pond or stream. Here the ecotones (i.e. transitional zones) between the two can also be vital as buffer zones and reservoirs for biodiversity, but difficult to assign categorically to either improved land or semi-natural habitat. Furthermore, the existence of partially improved semi-natural grasslands in Wales means that there are large areas that may be responsive to future management for either greater or lesser agricultural productivity (see ERAMMP Report 4: *SFS Evidence Review Annex 4 Building Ecosystem Resilience*). Consequently 'enclosed farmland' can contain a mosaic of habitats. In many cases however, the enclosed land of a farm will consist of more

[1] ERAMMP Report 4: SFS Evidence Review Annex 4 Building Ecosystem Resilience https://erammp.wales/en/r-sfs-evidence-pack

‘improved land’ than ‘habitat land’, reflecting the principal goal of maximising agricultural productivity.

Improved land has both existing and potential value for biodiversity conservation. There are widely recognised wildlife communities associated with improved farmland, including wild species that depend either wholly or partially on such land, for example threatened arable weeds, and birds such as skylark (dependent on open-field arable fields or extensively managed grassland). Conservation of such communities has been recognised as a priority in Wales, the UK and across Europe for several decades, and a wide range of management interventions aimed at species conservation or habitat condition on improved land have been supported under agri-environment schemes. In addition, ‘farmland’ is, in reality, a matrix of semi-natural and improved habitats, often with land cover units that are small relative to the scale of habitat dependence of key species. Semi-natural habitats can also interact with improved land, for example when species of bird or bee nest in a semi-natural habitat and forage in improved land, or where adjacent woodland makes open fields unsuitable for species such as skylark or lapwing. Therefore, it is important to consider at multiple spatial scales how target biodiversity is influenced by both management and the interactive effects of improved land and semi-natural habitats: field- or plot-scale patterns do not necessarily scale up to landscape scale - and some responses may only be meaningful at landscape scale.

1.2.2 Ecological resilience

This review is specifically concerned with resilience of ecosystems. ‘Resilience’ has been used in several different ways in the ecological literature and the definition used here is taken from SoNaRR: "*the capacity of ecosystems to deal with disturbances... whilst retaining their ability to deliver services and benefits now and in the future.*" The second part of this definition potentially conflates ecosystem service provision with ecological condition, in the sense that it is possible to take a view in which habitats are valued only insofar as they provide benefits for humans. But it is equally possible to take a wider view by also considering conservation as having value for its own sake. Accordingly, most research regarding interventions and their effects on biodiversity involves benefits or otherwise for plant and animal species and habitats themselves.

Deriving evidence of resilience is challenging. There are few demonstrations of the links between ecosystem properties before and after a perturbation that can be used to infer resilient behaviour. This is because such evidence depends on correlating higher and lower levels of those attributes expected to confer resilience with measurements of the response to perturbations. Moreover, there are many types of perturbation to be considered, such as extreme weather events or acute environmental challenges such as pollutant leaks. The unpredictability of these events makes data collection very difficult logistically, and therefore resilience is often assessed using proxies that are justified theoretically. Examples include habitat connectivity supporting movement and recolonisation of a site, and hence buffering extinction events; and species diversity or functional redundancy conferring resilience of function in the event of the loss of particular species. However, such approaches inevitably involve assumptions that have varying degrees of evidential support, such as the effectiveness of a given habitat connection in promoting movement, or the accurate assignment of equivalence of function to multiple species.

1.2.3 Habitat Condition

This review follows the brief in presenting evidence relating to habitat condition, and only considering the presence, abundance or diversity of species where these are constituents of habitat definitions or condition metrics. It is important to note that the relationship between habitat condition and species presence or abundance is largely unknown and likely to vary with habitat context, which means that one cannot simply assume that species associated with a habitat will necessarily benefit from measures targeted at improving the condition of that habitat (or vice versa). However, it has been demonstrated in a wide range of contexts, including in Wales[2] that species respond to management interventions on farmland.

1.2.4 Linking evidence to farm types

As defined in SoNaRR, enclosed farmland in Wales covers around 1 million ha or just over half the total area of agricultural land. It is managed predominantly as agriculturally improved grassland, characterised by the dominance of rye-grass and clover, and is used for dairy, beef and sheep production (the latter often in association with use of extensive semi-natural pastures in the ffridd and uplands). The area of arable and horticultural crops in Wales increased by a third between 2006 and 2016 but accounts for less than 9% of the enclosed farmland.

In this review the evidence for 'in field' interventions on improved land has been structured by arable and grassland interventions, rather than by farm type. For other interventions the arable/grassland distinction is unnecessary, for example in the management of agriculturally unproductive land and features across the farm. Landscape-scale evidence is, by definition, integrated across a number of different farms and/or extensive mosaics of different land cover and habitats, which further weakens the usefulness of linking evidence to specific farm types.

1.2.5 Uncertainty

It is important that any review of the evidence behind interventions takes proper account of uncertainty. This may be in the form of statistical uncertainty in analyses of national-scale responses, but may also be more qualitative, such as the application of spatial comparisons to infer the effects of temporal change, or the assumption that laboratory-, patch-, field- or farm-scale evidence from small-scale short term trials will scale up to nationally-relevant effects over the long term. For example, a test on a pair of farms with contrasting management may have used a flawless protocol, analysis and interpretation, leading to clear and precise results, but there is inherent uncertainty in assuming that such results predict national-scale responses over the period of interest. This is particularly important to recognise because it is much easier to design and to conduct experiments or controlled trials over short periods and small scales, while long-term, large-scale studies are inevitably subject to more noise and error. The former will, therefore, tend superficially to provide 'better evidence', while masking the inherent uncertainty in the wider representativeness of the patterns that are revealed. There is also high variability in the extent of the research that has been conducted into different taxa. For example, birds are well-studied at the management option or field scale, and also at the landscape scale, whereas the effects of management options on moths have only been considered for hedgerow management and grass buffer strips.

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[2] For example, analyses conducted under GMEP: Dadam & Siriwardena (2019)

Furthermore, most generic bat monitoring is probably insensitive to the kind of population change effects that are sought in agri-environment scheme evaluation studies, but only a few such studies having been conducted.

Whole-farm and agreement-level comparisons

Agri-environment scheme interventions are typically put in place as part of an agreement or contract that covers an entire land-holding, so the effects of individual management options occur in the context of a wider portfolio of interventions. Moreover, policy evaluations are often made at the scheme level and policies can encourage, or dictate, how options are used in combination (Winspear et al. 2010). Monitoring and evaluation studies have therefore often considered whole farms as management units, providing evidence about the effects of schemes as a whole, but not allowing the separation of the effects of different option types. These studies are clearly relevant to the evidence base for agri-environment scheme impacts, but do not fit readily into an evidence review of individual intervention types (unless entire agreements comprise just one type of intervention).

Research on birds considering whole agri-environment scheme agreements at the farm level has mostly considered only “narrow-and-deep” agri-environment scheme management, i.e. locally intensive management at the farm scale (as opposed to “broad-and-shallow” management spread more thinly across a landscape), with either a single-species or a farmland bird-community focus. Management for cirl buntings in south-west England has taken an inclusive approach such that the old Countryside Stewardship and Higher-Level Stewardship (HLS) agreements implemented there have integrated options to provide all the species’ requirements, as far as possible. Surveys of species-specific breeding and wintering bird abundance, to reveal habitat use and population changes, have shown strong increases in farm-level densities that are attributable to the agri-environment scheme management, but the management has focused on the conservation outcome, rather than on testing individual option effects, making attribution to individual options difficult (e.g. Peach et al. 2001). Thus, the recovery of the species could indicate synergies between different options providing different resources or, more simply, could show the effects of the limiting option among a broader suite of options.

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

Redhead et al. (2018) found variable patterns between years in spatial responses of multiple bird groups and species to combinations of options mimicking ELS and HLS agreements, but general positive effects of the AES treatments in all years, compared to controls. There was a tendency for the more intensive HLS treatment to have a larger effect, but this was far from clear. This study also found a strong dependence on wider landscape factors, but is limited by low replication: a single experimental estate, although large in area, and four, smaller, nearby control sites.

Larger, more mobile species are likely to be influenced by a wider range of different habitat features and the management that affects these habitats, suggesting that the potential effects of all such elements of management need to be integrated for comprehensive evaluation. This might include management to provide nest sites, summer food, cover, hibernation sites and/or winter food, for example. Applying this in an evaluation context, Perkins et al. (2011) showed that agreements with species-specific targeting and advice to farmers delivered local population increases in the corn bunting *Emberiza calandra*.

There is also uncertainty in the transferability of evidence from different contexts, for example dependence on evidence from outside Wales, even if this is from elsewhere in the UK where pedo-climatic conditions may be very different, for example in the main arable areas of eastern England and Scotland. Furthermore, uncertainty will often be particularly unclear in the case of collated expert judgement.

All of the above means that there will be a sliding scale in the confidence that can be attached to the evidence that a given intervention will work (or not), for example on its own or in synergy with other interventions, in scaling up of effects from field or farm to populations/landscapes, or moving from one location to another.

We therefore highlight the importance of the ‘amber’ category in Table 8.1 below and Table 3.1.1 in ERAMMP Report-10A *Sustainable Farming Scheme Evidence Review Integrated Analysis*[3] which captures interventions where the expert community agree there is an intervention logic chain which can be supported, but evidence is currently limited.

[3] www.erammp.wales/10a

2 Outcomes

The principal outcome of the interventions on improved farmland reviewed in this report is broad-scale increases in management that is relevant to biodiversity at both the farm and landscape scale. The interventions are aimed at improving ecological resilience, through enhanced condition, diversity, extent, connectivity and adaptability of biodiverse grassland, crops and non-productive habitats and features.

Habitats do not function in isolation from each other, and the scale, distribution and connectivity of different semi-natural features plays an important role in the functionality and resilience of habitats and species associated with farmland. In the review, this aspect has been considered at a landscape scale but it also applies at farm level. For example, in Wales the livestock systems have historically been associated with the inter-dependent management of improved land with different types of semi-natural habitat both on individual farms and at a landscape scale e.g. use of common grazings and away-wintering (*hafod y hendre*). Although these links have been weakened to some extent they remain very relevant to the implementation of the Sustainable Farm Scheme (SFS).

The Welsh Government brief acknowledges that unimproved semi-natural habitats and improved land exist as a mosaic across all farmland in Wales, rather than as isolated features. The existence of this mosaic at differing scales in the lowlands and uplands means that the outcomes of this review must necessarily be considered together with those of ERAMMP Report 4: *SFS Evidence Review Annex 4 Building Ecosystem Resilience*, as noted in 1.1.

The integration with ERAMMP Report 4 is particularly important in the case of two issues:

- The presence of *partially* improved semi-natural grasslands in Wales, within both enclosed farmland and the ffridd, raises a policy question about the future ‘direction’ in which these should be managed to deliver the SFS outcomes. Put simply, this is a choice between managing a semi-improved grassland to restore it to a semi-natural grassland or managing it to increase biomass production (e.g. of fodder, biomass crops or timber). In making this choice on individual sites the impact on biodiversity habitats and species at the landscape scale should be factored into the decision-making process.
- Improved or semi-improved land will in most cases be the starting point for the creation of new semi-natural habitats and woodlands on farmland, including connecting existing semi-natural habitats to conserve them and to improve their biodiversity status.

The most appropriate choice in each case will depend on both ecological resilience and economic considerations.

Depending on the type, scale and location within the landscape of the intervention, secondary outcomes for interventions on improved land could include carbon sequestration, reduction of water pollutants and new non-agricultural income streams, such as those from new agroforestry/woodland or private recreational use (e.g. horse-riding routes).

We recommend that different types of SFS contracts should be used to achieve biodiversity gains through environmentally sustainable management of improved land. A combination of annual and multi-annual contracts will be required for biodiversity management actions and/or outcomes, because some habitat improvements will benefit from longer contracts than is current practice, plus capital grants for associated actions (e.g. fencing, soil management). It will be important to ensure the longevity of biodiversity improvements, especially if it would be easy to revert to farming practices that would negate the biodiversity gains.

3 Policy Relevance and Policy Outcomes

These outcomes could contribute significantly to the following Natural Resources Policy priorities:

- Resilient ecological networks
- Maintaining, enhancing and restoring floodplains and hydrological systems to reduce flood risk and improve water quality and supply; (including catchment management approaches, natural flood management, soil management etc.)
- Restoration of our uplands and managing them for biodiversity, carbon, water, flood risk and recreational benefits (directly, where improved land extends significantly onto higher ground; and indirectly, where productivity gains on high yielding lowland soils allow ‘sparing’ of upland areas).

They also have the potential to contribute to additional priorities. This will depend on the implementation choices made on the objectives, characteristics and location of management of existing and creation of new non-productive habitats and landscape features.

- Increasing green infrastructure in and around urban areas

4 Introductions to the interventions reviewed

4.1 Evidence of farming trends, the implications for habitats and species and the need for interventions

Many of the species that have declined in recent decades in farmed landscapes are wholly or partly associated with the farmed land itself. These species use open fields for all or part of their life cycles and have massively declined in abundance or been lost from localities due to agricultural intensification. Their ecologies mean that these losses are not accompanied by a notional retreat to or concentration in semi-natural habitats. Affected species comprise iconic farmland birds, arable and grassland plants and grassland invertebrates. This has led to the adoption of agri-environment interventions of various kinds across Europe, intended to deliver some form of reduction in the intensity of agricultural management, whether within fields or across holdings or landscapes, and thus provide resources to support the recovery of target habitats and/or species.

Despite more than 30 years of implementing agri-environment schemes in Wales, SoNaRR reports that during the latter part of the 20th century in Wales, highly productive grass crops for silage production largely replaced semi-natural pastures and hay meadows, and that in the ten years between 2005 and 2015, within the improved grassland category, new leys (those less than five years old) increased by 37%. A move towards keeping larger, more productive sheep, accompanied by genetic gains within breeds, means that the sheep sector is producing more lambs from fewer ewes; these productivity gains between 2004 and 2012 were accompanied by more intensive utilisation of enclosed farmland[4].

One of the major consequences of agricultural intensification is to reduce the adaptive capacity of improved land to respond to a simple alleviation of management intensity. Altered composition of species pools, much reduced population sizes of remnant biota, plus high and persistent levels of nutrients mean that dispersal and establishment can take a long time (many decades), be highly uncertain and contingent on local conditions, and stabilize at low species richness (Critchley et al. 2000; 2004). Thus, interventions designed to achieve restoration objectives are less likely to work, or take much longer to work, where the legacy effects of agricultural improvement – high residual fertility, lack of responsive biota including seed sources and other modified abiotic conditions – are greatest (Critchley 2003; Oster et al. 2009; Donath et al. 2003; Pywell et al. 2002), However, a key caveat is that periods over which monitoring is typically carried out may well be too short to capture targets which could be achieved over a longer period. The evidence review below suggests that active steps to reduce nutrient surpluses and to introduce biota (particularly plant material) are required to accelerate and achieve significant recovery of biodiversity on improved land (e.g. Lawson et al. 2004; Török et al. 2018). However, the more ‘active’ these interventions are, the more costly they are per unit area (Török et al. 2011; Donath et al. 2007; Cuttle et al. 2016; Collins et al. 2018). Nevertheless, a range of taxa depend specifically on improved land that is actively farmed at varying levels of intensity, so there is no simple relationship between ‘naturalness’ and value for biodiversity.

[4] SoNaRR (2016) Chapter 3 Annex.

The legacy effects of agricultural intensification are also relevant when considering novel large-scale management. In recent years, landscape-scale intervention has captured the imagination of the public as well as polemicists and ecologists (Monbiot 2013; Jepson 2014). The idea is to restore a suite of natural processes, which are expected to impart resilience as a result of achieving renewed dynamism and adaptive capacity. These properties are expected to be optimised where connectivity is high, habitat areas are large, where species diversity is high but inevitably dynamic and varying from place to place, and where ecological conditions are in some sense 'better' than before. There is however little evidence on how the legacy effects of intensive agricultural management will interact with such large-scale attempts to reboot the dynamics of ecosystems. There are risks too. For example, regimes such as large-scale, low intensity grazing and natural flooding regimes could result in greater dispersal of residual nutrients and 'undesirable' plant species. While this might not drive further deterioration in already highly modified systems, these processes could put at risk existing patches of semi-natural habitat embedded within improved land.

Furthermore, the interaction between new, large-scale visions and the place-specific legacy effects of agricultural modification pose novel practical challenges for the design of interventions. For example, Standish et al. (2014) defined the notion of 'unhelpful resilience', where long managed, unnaturally productive systems and their associated biota prove resistant to attempts to move linearly back down fertility and disturbance gradients. Thus further evidence gaps centre on defining and measuring the dynamic reference conditions against which progress should be judged (Hiers et al. 2012; Hobbs and Harris 2001) and estimating the timescales required to achieve target states and their dependence on legacy effects of land-use in any one place. Detailed evidence of what works and what does not at the large-scale (and why) is lacking but will increase as lessons are learned from an increasing number of rewilding and large-scale restoration exemplars (Tree 2018; Lengyel et al. 2012).

4.2 Intervention categories

Links with other ERAMMP Evidence Reviews

This evidence review is a companion document to ERAMMP Report-4: *SFS Evidence Review Annex-4 Building Ecosystem Resilience*[5] which covered semi-natural habitat management and restoration on farmland outwith improved land. The biodiversity management of farm trees, hedgerows, farm woodlands and agroforestry is covered in that *Review* and, in the case of woodlands and agroforestry, also in the forthcoming *ERAMMP Report-32: National Forest in Wales – Evidence Review*[6].

Other 2019 Evidence Reviews considered some of the interventions on improved land from a different perspective (e.g. sward diversification[7], soil nutrient management plans[8] and wetland and water management for flood risk mitigation[9]).

[5] https://erammp.wales/en/r-sfs-evidence-pack
[6] https://erammp.wales/en/r-forest-evidence
[7] ERAMMP Report-2: SFS Evidence Review Annex-2 Sward Management: www.erammp.wales/2
[8] ERAMMP Report-3: SFS Evidence Review Annex-3 Soil Carbon: www.erammp.wales/3
[9] ERAMMP Report-9: SFS Evidence Review Annex-9 Flood Mitigation: www.erammp.wales/9

Selecting interventions for this review

Three criteria were used to select interventions on improved land for review in this report:

- biodiversity interventions appropriate for improved land that have already been reviewed in the ERAMMP Evidence Reviews on semi-natural habitats are the National Forest are flagged and cross-referenced clearly, but not repeated here.
- interventions should be applicable at a scale relevant to the majority of improved land in Wales, bearing in mind the relative scarcity of arable land (and also its biodiversity value because of that);
- interventions are likely to be above the future regulatory and contractual environmental baseline for farm payments for the SFS;

Intervention categories have been divided into five broad groups for the assessment of the evidence.

In-field management of the whole crop/grassland (Section 5.1)

Organic farming (Section 5.2)

Modified management of strips/plots around or within the field (Section 5.3)

Management of agriculturally unproductive land and features (Section 5.4)

Other interventions (Section 5.4)

The review then considers how the implementation and targeting of these interventions, individually and in conjunction, affects their impact on farmland biodiversity and the resilience of ecosystem services:

Significance of context, synergy and scale of biodiversity interventions on improved land (Section 6)

4.3 Evidence to be explored

Interventions can and should be put together in different combinations, depending on the specific local circumstances. Given the range of different potential interventions on improved land in Wales the focus is more general, addressing the apparent conservation needs of the categories of interventions listed above and the evidence for broad interventions that respond to these needs. Evidence will be broadly of two types (although not presented in separate categories):

A. evidence of the *need* for interventions, considered from two angles:
    i. habitat extent and condition, and the condition of relevant species populations; and
    ii. farming trends that influence this condition
B. evidence of the *effectiveness* of specific management interventions on semi-natural habitats.

Much of the available evidence is drawn from studies outside Wales, especially in the case of arable land. The limitations of different types of evidence is discussed in Section 1.2.5. Evidence from all relevant farming systems has been considered, but it is important to note that the relevance to the specific conditions in Wales varies. For example, most evaluation of agri-environment scheme intervention has involved arable systems, which are considerably rarer in Wales than in England, and the baseline management of grassland in Wales is less intensive than that in some other countries. This means that the quantity of evidence available, and presented in this review, is frequently not proportional to the practical significance of a particular intervention in Wales.

5 Interventions

5.1 In-field management of the whole crop/grassland

These interventions modify the existing management of the crop/grassland within the field in a way that may have benefits for biodiversity (including soil biodiversity) and vegetation cover.

5.1.1 Grassland in-field management

The grassland in-field interventions are considered under the following headings:

- Reduce fertiliser, pesticide or herbicide use; use organic rather than mineral fertilisers
- Adapt mowing or first grazing dates on improved or semi-improved grassland; use mowing techniques to reduce mortality; leave uncut patches in silage fields
- Change grazing management, including mob grazing
- Manipulate silage mowing height
- Convert improved/semi-improved grassland to species-rich grassland

Reduce fertiliser, pesticide or herbicide use; use organic rather than mineral fertilisers

Chemical inputs and artificial fertilisers have been implicated as the direct or indirect causes of loss of multiple taxa in farmland, for example in the loss of invertebrates as unintended victims of pesticides (Sotherton and Holland 2002), the loss of invertebrate food resources for farmland birds (Boatman et al. 2004) and the encouragement of dense, fast-growing perennial grasses as a result of applying inorganic nitrogen, at the expense of other, less competitive grasses and flowering plants (e.g. Cleland & Harpole 2010). Hence, an obvious management intervention is to reduce inputs. Note also that the use of livestock slurries, digestate or poultry manure could have similar consequences, as they have high available nitrogen contents. Moreover, they will have greater potential ammonia emissions than ammonium nitrate fertiliser. These are therefore not alternatives to inorganic fertilisers that would solve the problems described above.

An important consideration in the Welsh context is that much of the evidence for effects of reduced inputs comes from England, where grassland management may be more intensive, and that the evidence for reductions in inputs may involve both baseline and reduced levels that are above the baseline levels in Wales. Reductions in inputs from a lower baseline may not have the same effect. Nitrogen application on all grassland systems averaged 57 kg ha-1 across Great Britain in 2018, which was very similar to the 5-year average (Defra 2019). Specific figures for Wales are not available, but this figure is probably lower for Wales (see section 1.2.1). Grassland systems in Wales are also often deficient in phosphate from a production perspective, with application rates below the recommended agronomic amounts (Gibbons et al. 2014). In Wales the scope to reduce application rates of fertilisers without compromising productivity is therefor likely to be limited to the areas of improved land where nutrient surpluses do exist. At the national scale, it therefore may be more appropriate to focus on interventions that improve utilisation of the nutrients that are applied[10]. Such interventions have yet to be widely applied, however, so evidence for their efficacy for biodiversity is lacking.

Further, it is important to note that much of the evidence relevant to reductions in chemical inputs comes from studies of organic farming, so is conflated with a change at the level of an entire farming system, making it difficult to partition the specific effects of the lack of inputs. In this respect the evidence from monitoring agri-environment scheme schemes is highly significant (see below), but it should be noted that agri-environment schemes have typically aimed for an overall reduction in chemical use, so most options prohibit or restrict inputs alongside the main requirements (e.g. conservation headlands).

In their review of the performance of grassland management prescriptions in Environmentally Sensitive Areas (ESAs) Critchley et al. (2003) reported that successful reduction of soil extractable phosphorus (a decline from 14.3 to 10.7 mg l^{-1} over three years (Olsen extraction method)) in the hay meadows from the Radnor ESA coincided with achievement of restoration goals. However, the starting point here was semi-improved hay meadow. These changes coincided with reduced abundance of plant species favoured by high nutrient loads. Similar changes were seen in the Cambrian Mountains ESA hay meadows. Improved grassland starting points were much less responsive to restoration treatments with only one out of twelve instances being interpreted as achieving the restoration objective over seven years of monitoring. While semi-improved starting points were more responsive, even here, 12 of 17 instances failed to respond.

Some evidence suggests that a reduction in agrochemical inputs can benefit insect communities. For example, Albrecht et al. (2007a, b) showed that low-input grass in Switzerland was associated with enhanced pollinator bee communities in adjacent intensive pasture. Furthermore, Fuentes-Montemayor et al. (2011a) found that micromoth and macromoth species richness and abundance were higher in agri-environment scheme species-rich grassland than in conventional controls. Potts et al. (2009) investigated the effects on bees and butterflies of suites of grassland management options that included reduced fertiliser inputs. General extensification of management benefited butterflies, but bees only responded to sowing treatments that provided flower-rich habitat. Across Europe, a meta-analysis found positive effects of grassland extensification on pollinator abundance and species-richness, although effect sizes were larger for sown flower strips (Scheper et el. 2013). Low-input grassland in Switzerland has been shown to be associated with different butterfly communities to those in conventional grassland, across multiple spatial scales, but without an overall effect on species richness (Aviron et al. 2007). Conventional fields received an annual average of c. 200 kg N ha^{-1} and were cut around three times, whereas low-input fields received c. 7 kg N ha-1 and were cut twice. The differences probably reflect the communities of food plants present, but agri-environment scheme management effects also differed significantly with landscape context – notably the amounts of semi-natural habitat close to sampled fields (Aviron et al. 2007). Reductions or control of nitrogen applications have underlain popular grassland management options in English agri-environment schemes, but such options have been criticised as allowing inputs of up to 50 kg N ha^{-1}, which are greater than are typically applied to many fields, such that few additive management effects might be expected (Natural England 2008). Perhaps for

[10] See also ERAMMP Report 1: *SFS Evidence Review Annex 1 Soil Nutrient Management for Improved Land*

this reason, tests of the effects of this management have shown little evidence of effects on birds, although for options with tighter restrictions on nitrogen use there was also only equivocal evidence for positive and negative effects on population growth rates across bird species in the BBS (Baker et al. 2012, Dadam & Siriwardena 2019) These analyses of long-term agri-environment scheme effects have been conducted for England and Wales, mostly involving fertiliser and stocking reductions, but also changes to mowing regimes. They have shown mixed associations between grassland management and population change of target species, with a positive relationship for population growth rate of linnet *Carduelis cannabina* in Wales, but a negative one for skylark *Alauda arvensis* (results from analyses for Tir Gofal under the Glastir Modelling and Evaluation Programme (GMEP): Dadam & Siriwardena 2019). There is more evidence of associations between grassland management and growth rates in England, but these are also equally both positive and negative across species (Pringle et al. 2020). Overall, there is only limited evidence for these management options to have affected birds in practice, across the full range of species tested. The results of 'snapshot' field- and farm-scale sampling by MacDonald et al. (2019) similarly showed no effects of Tir Gofal in grassland on multiple taxa, and there was no evidence for selection of Glastir-managed grassland in preliminary analyses of bird field survey results from GMEP (GMEP BD011).

There were no significant effects of English agri-environment scheme grassland management options (as for birds above, combining mowing and input extensification measures) on brown hare *Lepus europaeus* population growth rates in an analysis of national survey data (Pringle & Siriwardena 2017). Barlow & Briggs (2012) found evidence that grey long-eared bats respond positively to grassland management, but Fuentes-Montemayor et al. (2011b) found no such effect for common pipistrelle.

Adapt mowing or first grazing dates on improved or semi-improved grassland; use mowing techniques to reduce mortality; leave uncut patches in silage fields

One consequence of the fertilisation of grass is that it can be cut or grazed earlier in the season than was the case traditionally. This provides grass or silage of higher quality (more digestible grasses of higher metabolisable energy and protein) for livestock, thus reducing the need for bought-in feed.

For bird (and probably other) species that breed in grassland, this can have severe consequences for mortality and breeding success because they are still present and/or immobile when mowing takes place. Greater mowing frequency than is viable in, say, hay meadows, can also leave insufficient time between cuts for bird nests to be established and for chicks to fledge. Therefore, delaying cutting, using specific mowing techniques[11] that allow mobile chicks to escape, marking nests to allow avoidance and leaving uncut strips as refuges, all have the potential to reduce these impacts.

There is strong evidence from across Europe suggesting that adapted mowing can be very effective in increasing breeding success of ground-nesting bird species by reducing mechanical nest or chick destruction (e.g. corncrake (*Crex crex*) in Scotland (Tyler et al. 1998), whinchat (*Saxicola rubetra*) in France (Broyer 2011)). However, a 13-year study of breeding wader densities in the Netherlands found that adapted mowing practices at best only maintained existing differences between treatment and control areas (Breeuwer et al. 2009). It was recommended that additional

[11] For example, by retaining (or mowing last) patches or strips along field borders

management to raise groundwater levels and to reduce fertilization would allow the development of an open vegetation structure and hence increase chick survival sufficiently to drive population increases. Nest marking to allow farmers to avoid nests during farm operations prevented direct nest destruction, but other causes of loss, such as predation, increased in compensation, leaving no net effect on wading bird breeding success (Kragten, Nagel et al. 2008). However, a similar study in the Czech Republic found that nest losses due to predation were not increased by marking, so the effectiveness of such measures may depend on the specifics of the approach taken or the context (Zámečník et al. 2018). As noted in the section above on input reductions, mowing management has been tested as an influence on national population growth rates from the BBS in England and Wales, within the broader set of grassland management options described above, but without showing clear effects for more widespread bird species (Baker et al. 2012, Dadam and Siriwardena 2019, Pringle et al. 2020).

Experimental grassland plots in England demonstrated that the benefits of sowing grasses and legumes for pollinator biodiversity tend to be greatest when there is a summer rest period in cutting and grazing management (Woodcock et al. 2014). However, the effect of the summer rest period was not present when sowing seed mixes with grasses only, or with grasses, legumes and forbs. Potts et al. (2009) investigated the effects on bees and butterflies of suites of grassland management options that included reduced mowing frequencies. General extensification of management benefited butterflies, but bees only responded to sowing treatments that provided flower-rich habitat. Wilkinson et al. (2012) considered the effects on pollinators of targeted agri-environment scheme management for the corncrake (*Crex crex*). The promotion of early and late tall grassland vegetation cover had significant positive effects on the abundance of four of five bumblebee species that were tested.

There is good evidence that small mammal abundance (i.e. of voles, mice and shrews) is higher in taller swards in agri-environment scheme grassland margins, i.e. those that are cut only every 2-3 years (Askew et al. 2007).

Change grazing management, including mob grazing

Lenient early season grazing followed by early cessation is predicted to change vegetation structure and, hence, invertebrate communities in grassland. This has been studied on 13 pasture fields in south-west England over four successive years, with a focus on food provision for birds (Eschen et al. 2012). Lenient early season grazing increased total invertebrate abundance by 71% after 4 years, the most responsive groups being Araneae, Auchenorrhyncha, Collembola and Heteroptera. Early cessation of grazing led to increased cover of the grasses *Holcus lanatus* and *Agrostis* spp. and increased litter accumulation, but reduced clover cover and sward structural heterogeneity. The changes may therefore have increased food availability to birds, but reduced accessibility to it. The evidence for the value of this management type to insectivorous birds is therefore limited and unclear.

Low intensity grazing of unfertilised grasslands across the UK, Germany, France and Italy has been shown to increase species-richness and abundance of butterflies and grasshoppers (Wallis De Vries et al. 2007). However, at the UK site, the encroachment of competitive grasses and a decrease in plant species richness also occurred under low intensity grazing (Scimone et al. 2007). A comparison of intensively and extensively grazed lands in North Germany found that extensively grazed sites had higher biodiversity of grasshoppers, butterflies, bees and wasps

(Kruess and Tscharntke 2002). These studies include grasslands which are on the boundary between semi-improved and semi-natural, but their findings are relevant for this review.

There is a growing trend for farmers to re-evaluate their grazing management practices to improve the utilisation of grass thereby reducing reliance on bought-in feed and make better use of applied nutrients. This has led to a rotational grazing approach being advocated over traditional 'set-stocked' approaches (e.g. La Canne and Lundgren 2018) and in some cases the adoption of 'mob grazing'. Mob grazing is a type of rotational grazing which takes different forms, but net impacts may include trampled swards, swards not being grazed to below 10cm and high densities of dung and urea inputs, depending upon the farmer’s choice of management. Management approaches may be highly individual to particular farmers, but in most cases, they are aiming to maximise grass productivity as well as potential wider ecological benefits. These approaches tend to involve shorter, more intensive grazing periods on fields, interspersed with longer rest periods (beyond a month). Therefore, total grazing pressure may be reduced, but often with short-term peaks at higher intensity than previous maxima. Many of these farmers are tying in changed grazing practices with sowing species-rich swards, under the banner of ‘regenerative farming’ (La Canne and Lundgren 2018). Expected net impacts on biodiversity would be those associated with longer swards and increased flowering of legume and herb species where present in the swards. Longer swards also provide additional habitat for some species of invertebrates and small mammals (see above) as well as potentially resulting in increased root density and associated soil diversity. New research is being conducted in these management approaches and their potential benefits for biodiversity in the UK.

Note that changing grazing management has not been investigated independent of other management for birds and mammals. Reduced grazing per se is likely to be positive, at least locally, for most species in improved grassland, but has only been studied in combination with interventions such as reduced inputs (e.g. Baker et al. 2012, Pringle et al. 2020). There have been no studies to date of the effects of mob grazing, for example.

Manipulate silage mowing height

A solution to the problem of insufficient time between silage cuts for ground-nesting birds to raise broods could be to provide refuge areas in fields in which birds are attracted to nest. This has been trialled for skylark (Buckingham et al. 2010), with a raised cutting height for the first cut of the season used on parts of fields, with the aim of delivering a preferred habitat that would attract new nesting attempts. However, a sufficient proportion of birds was not successfully attracted to these areas, so the trial was not successful, and the method has not been used more widely.

Convert improved/semi-improved grassland to (more) species-rich grassland

Introduction of plant species to grasslands, especially legumes, is often critical to achieve increases in the biodiversity of plants and invertebrates (Woodcock et al. 2013). 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). 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). To maintain sward diversity, it may also be necessary to reduce soil fertility which can be done by soil stripping or appropriate grazing or cutting management (Bullock et al. 2011). However, ERAMMP Report-2: *SFS Evidence Review Annex-2 Sward Management* notes that most of the studies of the diversity-function relationship in improved grasslands have been carried out in ungrazed systems, with few in grazed fields. In the case of agriculturally semi-improved grasslands that retain their potential for habitat improvement (towards species-rich grassland), changing grazing management is part of a suite of interventions reviewed in ERAMMP Report-4: *SFS Evidence Review Annex-4 Building Ecosystem Resilience.*

5.1.2 Arable land in-field management

The arable land infield interventions are considered under the following headings:

- Reduce fertiliser, pesticide or herbicide use
- Use organic rather than mineral fertilisers; input other organic matter
- Reduced tillage
- Introduce or modify crop/grassland rotations; companion cropping; undersow spring cereals; and cover/catch crops (including legumes)
- Leave overwinter stubbles, unsprayed, into late winter
- Leave overwinter stubbles unsprayed and follow with a spring fallow
- Fallow/unsown plots
- Arable reversion and landscape heterogeneity

Reduce fertiliser, pesticide or herbicide use

Reduction in fertilisers, pesticides and herbicides (plant protection products) in arable systems takes place along a gradient, with organic farming (discussed in section 5.2) at one end, where the use of pesticides or herbicides is prohibited, and only organic fertilisers are used. As with grassland, reduced inputs are expected to reduce direct mortality among beneficial and neutral invertebrates, to enhance invertebrate food availability for insectivores and to reduce crop growth, opening up access to soil and light for incidental animal and plant species.

Management options can include partial reduction in fertiliser, pesticide or herbicide use, rather than an outright cessation of use. For example, this could be a consequence of the use of integrated pest management and decision-support systems to predict disease and pest outbreaks, and precision agriculture to reduce the use of inputs and minimise the impacts (Keulemans et al. 2019). It could also come from combining a number of options to reduce the need for additional inputs e.g. reduced tillage is often associated with cover cropping (Chenu et al. 2019), and with crop rotation which could help to increase beneficial arthropods, improve soil quality and control pests (Schipanski et al. 2017, van der Putte et al. 2010). Much of the evidence on the impacts of reductions in fertiliser and plant protection products comes from organic agriculture, or from observations of the impacts of high use and assumptions that these will be reversed when this level of use is ceased or reduced. There is less evidence of the impacts of reductions, particularly in relating the degree of reduction to impacts (Keulemans et al. 2019).

Herbicides reduce the abundance of weeds, and in doing so they also reduce species dependent on them e.g. invertebrates and birds (Chiverton and Sotherton 1991; Smart et al. 2000). They can also affect plant nutrient levels and hormone pathways used in defence, which may influence plant susceptibility to invertebrate herbivores so that host suitability changes (Egan 2014). Reduced use of herbicides is likely to increase invertebrates. Floral resources for pollination should also increase, increasing the abundance and diversity of pollinators (Pywell et al. 2015).

There have been increasing concerns about the impact of pesticides on pollinators, particularly bees (Williams, Troxler et al. 2015; Ollerton et al. 2014). Studies have found that pesticides (such as pyrethroids, neonicotinoids and fungicides) whilst not individually impacting on pollinators, can interact to have colony level effects (Sanchez-Bayo and Goka 2014). Laboratory and field experiments have shown that insecticides can negatively affect moths in arable field margins (Hahn et al. 2015) while wild bee declines in England are correlated with pesticide use (Woodcock et al. 2016).

Reduced use of pesticides can encourage soil biota, in particular, allowing for increased earthworm numbers (Pelosi et al. 2014, Zwart et al. 1994).

Reduced pesticide use is expected to increase invertebrate food availability for birds (Campbell et al. 1997), but the evidence that this leads to significant effects on populations is limited.

Use organic rather than mineral fertilisers; input other organic matter

Organic fertiliser (animal manure) is mainly derived from cattle, pig and poultry, farming with liquid manure (slurry) having a lower dry matter content than solid manure (Bernal et al. 2009). In general, slurry has lower concentrations (g kg^{-1} fresh weight) of C and N than solid manure, but pH stays largely unaffected. Poultry manure (solid or liquid) generally has higher C and N concentrations than cattle and pig manure (Bernal et al. 2009). Green manure refers to crops that are grown specifically to be incorporated in the soil as a source of organic matter (cover crops may also be grown as a source of green manure, as well as providing vegetative cover to reduce the risk of soil erosion). Note that organic fertiliser sources may not deliver lower nutrient inputs than inorganics and can also be worse sources of emissions, as described for inputs to grasslands in section 5.1.1.

Other types of organic inputs to arable soils include compost, digestates, biosolids recycled from sewage (Maskell et al. 2019) and incorporated crop residues, such as straw. Biochar[12] is another form of organic material that can be added to soil, although this is not common in the UK.

These options tend to be implemented because of their potentially positive effects on soil organic carbon (Alison et al. 2019, Jones et al. 2006, Chenu et al. 2019), soil quality, structure and water holding capacity (Freibauer et al. 2004). Benefits to biodiversity above-ground are more likely to be linked to systemic changes (such as the adoption of fully organic regimes) than to replacing mineral fertilisers with organic fertilisers, because the negative effects of fertilisation per se will occur regardless of the specific source of crop nutrients.

Reduced tillage

[12] Biochars are obtained by thermal treatment of organic material in low oxygen conditions (Qambrani et al. 2017) and can be a side-product of liquid biofuel production.

Conventional tillage uses a mouldboard plough and multiple trips across the field with other tillage tools (e.g. discs and field cultivators) to invert the soil. Reduced tillage replaces the mouldboard plough with a lighter tillage implement like a chisel plough which disturbs less of the surface crop residue, and often only involves single passes with a disc and/or field cultivator. In no-till systems, a self-contained planting unit is used to plant the crop in a single pass with no seedbed preparation (Harper et al. 2018). Reduced tillage reduces the potential for erosion losses of bare soil, and has been recommended for carbon sequestration (Lal 2004) although there is limited evidence for its efficacy in a UK context (Moxley et al. 2014). Physical disturbance during tillage disaggregates and aerates the soil, accelerating soil organic carbon decomposition (Mikha and Rice 2014), and reduced tillage was thought to prevent this. Reduced tillage also encourages the maintenance and formation of soil aggregates. This is a desirable property that has been linked to carbon storage and resistance to erosive forces (Six et al. 2004; Duchicela et al. 2013) and there is an economic argument to focus on the avoidance of soil compaction and erosion on intensively farmed soils (Graves et al. 2015). Increased soil organic carbon in the topsoil changes the soil structure, enhances soil quality and reduces soil erosion, and there may be other beneficial effects including reduced energy usage, soil structure improvement, better aggregation, conserved soil moisture and improved water infiltration (Maskell et al. 2019).

However, the evidence for impacts on soil organic carbon are mixed and vary by soil depth. There are also risks of increased N_2O emissions (see ERAMMP Report3: *SFS Evidence Review Annex 3 Soil Carbon Management*).

Although reduced tillage may increase beneficial pest predators and so control arthropod pests (Schipanskia et al. 2017, Heroldova et al. 2018), the lack of soil inversion usually increases weed infestations and changes the composition and functional attributes of the weed community (Carmona et al. 2015, Armengot et al. 2016, Weber et al. 2017). There may be a need for increased herbicide use which can lead to herbicide resistance (Harper at al. 2018), but using crop rotations and cover crops in conjunction with reduced tillage can help to control weeds (Weber et al. 2017, Carr 2017, Creamer et al. 2002)

Reduced tillage increases the ecological sustainability of agroecosystems by maintaining high populations of soil-ameliorating fauna and insect pest predators. This may increase the small mammal density and diversity, including those of insectivores, as has previously been confirmed in set-aside fields (Heroldova et al. 2018).

Evidence for effects of reduced tillage on birds is inconclusive. In winter, skylarks *Alauda arvensis*, granivorous passerines and gamebirds have been found to occupy a greater proportion of cereal fields that were established by non-inversion tillage than conventional tillage in Britain (Cunningham et al. 2005). However, a further trial in Hungary found that the patterns varied between winters, with a range of species selecting ‘conservation tillage’ fields in year one, but only one (starling *Sturnus vulgaris* or skylark) in each of years two and three (Field et al. 2007). Although some research in North America has suggested that bird densities are higher in reduced tillage systems, the evidence for bird food resources for breeding birds in Britain is mixed (Cunningham et al. 2004) and bird abundance or population responses have yet to be investigated, although skylarks have been found to breed earlier in conservation tillage fields, which is expected to provide a breeding success benefit (Field et al. 2007). Specialist farmland species have been found to be less abundant in conservation-tillage farms than in conventional ones, with complex effects of foraging guild (granivore/insectivore/omnivore), and an interaction with the time fields have been under conservation tillage for granivores (Ondine et al. 2009). One reason for inconsistent results for effects of reduced tillage systems is variation in the alternative method of weed control, as mechanical and herbicidal approaches have very different effects on birds, and herbicide can be more significant as a negative influence than tillage (Barré et al. 2018).

Introduce or modify crop/grassland rotations; companion cropping; undersow spring cereals; and cover/catch crops (including legumes)

Crop rotation interventions include the use of multiple crop species/varieties in a rotation, integration of short-term grass or other non-woody, perennial leys into previously arable-only rotations, and alternating spring and winter crops to manage weeds (used particularly in no-till systems).

Above ground biodiversity can be enhanced by the crop species selected, which provide a range of habitats, and differences in flowering times can attract a wider variety of pollinators (Defra 2018). For example, early mass flowering oilseed rape has some beneficial effects for bumblebee colony growth, although this needs to be complemented by food availability in the late season (Westphal et al. 2009). More diverse crop rotations can increase soil microbial richness and diversity (Venter et al. 2016), possibly as a result of different organic matter inputs and changes in soil structure. A wide variety of crop types can promote soil health by improving the diversity of root architecture and reducing disease/pest burdens (Defra 2018).

It is well accepted that crop rotations can disrupt pest and disease cycles (Smith et al. 2008), potentially reducing the need for pesticide application. More diverse crop rotations can have lower weed densities (Schipanski et al. 2017, Cardina et al. 2009), particularly if legumes are incorporated into the rotations. However, many ‘weed’ species are also highly desirable from a biodiversity perspective (e.g. Smart et al. 2000). Perennial forage crops are more effective than annual crops at suppressing annual weeds in the following crop, and can shift the composition of weed communities over time (Schipanskia 2017, Entz et al. 2002). Crop rotations can be used to combat herbicide resistant weeds and can break the lifecycle of host specific pests and pathogens (Marcroft et al. 2004). Beneficial insect species also respond to previous catch or winter cover crops (Lundgren and Fergen 2010). Predator abundance was greater in maize following an autumn-planted, spring-killed grass cover crop, compared to maize without a preceding cover crop (Lundgren and Fergen 2010).

Including legumes in a crop rotation can improve long-term soil fertility and reduce fertiliser costs. Perennial crops typically have larger root systems than annuals, and integrating them in the rotations can contribute to soil quality improvements by alleviating soil compaction, reducing soil erosion due to reduced frequency of soil cultivation, and facilitating drainage (Zan et al. 2001, Lynch and Wosciechowski 2015).

The effects on birds of introducing new crop rotations have not been investigated, but the landscape effect of such interventions would be to increase land-use heterogeneity, which is likely to benefit both a range of individual species and community diversity.

Cover crops are fast maturing crops grown within a rotation (after harvest) to maintain soil cover during fallow periods (Defra 2018), and are typically ploughed under as green manure, or killed with herbicides under no-till systems. Legumes may be introduced to break a long arable run, or grown with the arable crop (intercropped or intersown). The vegetative mulch produced when cover crops are killed can be very effective at suppressing weeds when retained on the soil surface (Carr 2017, Dorn et al. 2015). To support weed management, it is best to use locally adapted cover crops that have rapid establishment, good soil coverage and high dry matter production (Dorn 2015).

Cover crops can have a positive effect on the biodiversity of many taxa especially invertebrates (Holland & Luff 2000, Lundgren & Fergen 2010). Soil biological activity and diversity are higher in systems with a surface mulch or cover crops (De Aquino et al. 2008). Legume crops can encourage on-farm biodiversity by providing a habitat for microorganisms and invertebrates (Veronesi et al. 2011). Birds are not a target for cover crop benefits and the effects have not been evaluated, but they are unlikely to be positive: wintering birds that use field habitats generally require access to bare ground to forage for seeds or invertebrates, and green cover over the crop stubble impedes this, although there could be benefits for species that forage under vegetation for invertebrates which exploit the microclimate created by the cover. The extent of any net effect on bird communities in any area will depend on the availability of other winter foraging habitat locally; if there are other stubble or ploughed fallows nearby, most birds are likely to relocate. Cover crops are one of a suite of agri-environment scheme options that are expected to provide winter cover for brown hare, and this suite was associated with positive effects on population growth rates in an analysis of national survey data (Pringle and Siriwardena 2017).

Undersowing is a traditional practice consisting of creating a grass ley in a rotation after a cereal crop. Effects on biodiversity have received little research attention, but there is evidence that undersown stubbles had slightly fewer weed species than those stubbles that were not undersown (Robinson & Sutherland 1999), although they support greater densities of sawfly larvae in Spring, providing an important food resource for grey partridge chicks (Barker et al. 1999). There have been no trials of this management mechanism as a conservation measure, but historical anecdotal reports emphasise its value (Shrubb 2003, O’Connor and Shrubb 1986, Potts 2003).

Leave overwinter stubbles, unsprayed, into late winter

Leaving crop stubbles over winter is a traditional component of arable management ahead of spring sowing, but one that declined with the availability of autumn-sown cereals and break crops (notably oilseed rape and field beans), which reduce weed competition problems and the uncertainties associated with spring weather. The loss of stubbles has been widely implicated as a cause of biodiversity loss on farmland because stubbles (and, to a lesser extent, ploughed over-winter fallows on heavy soils) are high-quality winter foraging habitats for birds, primarily due to the presence of weed seed and spilt grain, provided that herbicides are not applied post-harvest. Stubbles are also heavily used by brown hare. Support for unsprayed stubbles is a feature of all UK agri-environment schemes. In practice, this will guarantee a lack of herbicide input to existing stubbles preceding a spring crop and the retention of the stubble into late winter (typically February). However, It is unlikely to cause a change in crop rotations whereby spring cropping increases. The agri-environment management is expected to increase over-winter survival of farmland birds via enhanced food availability and, hence, promote population growth.

Cereal stubble management has been well studied and includes studies of habitat selection by birds in winter at the field and farm scales, as well as of responses of breeding populations at the national scale. Cereal stubble is recognised as both a preferred foraging habitat for granivorous birds in winter (e.g. Buckingham et al. 1999, Gillings and Fuller 2001) and a significant correlate of population growth rates via positive effects on over-winter survival (Gillings et al. 2005). Agri-environment scheme stubble options critically preclude herbicide inputs, to enhance weed seed availability, and ensure stubbles are left unploughed until late winter, so studies of the value of stubble *per se* are not relevant to assessment of agri-environment scheme impact. There is some evidence that unsprayed stubbles actually attract birds more than conventional, sprayed stubbles (e.g. Gillings & Fuller 2001), but the patterns are not always clear-cut (e.g. Field et al. 2011), possibly because weed seed banks in some fields have declined to the extent that a lack of spraying does not allow a rich flora of seed-bearing plants to be established (Robinson & Sutherland 1999). Long-term, national-scale monitoring data have been analysed to measure the effects of stubble management and other agri-environment scheme options on bird population growth rates in England and Wales (including work under GMEP: Dadam & Siriwardena 2019). There is strong evidence that stubble management has a positive impact on farmland bird population growth rates (in one or more landscapes and at one or more scales), probably reflecting the provision of winter food resources that address the key limiting factor for abundance (Baker et al. 2012, Dadam & Siriwardena 2019, Pringle et al. 2020).

Stubble management is one of a suite of agri-environment scheme options that were expected to provide winter cover for brown hare, and this suite was associated with positive effects on population growth rates in an analysis of national survey data (Pringle & Siriwardena 2017).

Leave overwinter stubbles unsprayed and follow with a spring fallow

Traditional crop rotations often included year-long fallows, where fields were left uncropped from harvest in year one until sowing for year three. These fields developed rich floras of weeds and crop volunteers, with diverse vegetation structure and high accessibility of bare ground. Fields that were ploughed and then fallowed provided many of the same resources. They became less common with agricultural intensification, as chemical inputs replaced and surpassed the yield benefits of fallowing, but re-appeared as rotational set-aside under the Common Agricultural Policy (CAP) in the 1990s. The latter fields were left without inputs from harvest or ploughing until the end of following cropping season, providing the winter benefits of an unsprayed stubble, followed by a structurally diverse, plant- and invertebrate-rich spring habitat. Following the demise of set-aside in 2008, options recreating this habitat management were added to agri-environment schemes such as Environmental Stewardship in England.

To date, there has been little specific monitoring of an agri-environment scheme option of “extended stubble management” with summer fallow, although the winter effects of the option should be the same as those of the standard overwinter unsprayed stubble option (discussed above). A two-year study on 28 farms piloting management under the Arable Stewardship Pilot Scheme in England showed that lapwing *(Vanellus vanellus)* nests in these fields had greater daily survival probabilities than nests found in other arable fields. To date, effects of this option on population growth rates of target species have been mixed and unclear, but rates of option uptake are not high and effects may become clearer over time (Pringle et al. 2020). The evidence for positive effects of this specific management on birds is, therefore, limited to the results of studies of rotational set-aside, which show strong evidence of habitat selection by a wide range of bird functional groups in both winter and the breeding season (Henderson et al. 2000 a,b).

Extended stubble was associated with positive effects on brown hare population growth rates in an analysis of national survey data (Pringle and Siriwardena 2017).

Fallow/unsown plots

A feature of intensively managed crops is high vegetation density and tall plants. This restricts access to suitable bare ground for nesting birds and can force them to nest at low density and in locations such as spray lines, where they are vulnerable to predation and to destruction during field operations. A crop management solution has been to provide small (c. 4 m^2) or large (c. 1 ha) fallow plots in fields, which are aimed at skylark (*Alauda arvensis)* and lapwing (*Vanellus vanellus)*, respectively, created by avoiding sowing the crop (or spraying it off after germination). Birds can nest in these plots or use them to enter the crop itself. Trials demonstrated positive effects of the smaller plots on skylark breeding success.

Overall, the farm-scale evidence suggests that skylark plots have positive effects on breeding success and territory density, although benefits have been shown to be compromised by co-location with grass margins because benefits for breeding success are more than negated by apparent facilitation of nest predation (Morris et al. 2004). Results from Sweden support the positive patterns found in England (Eggers et al. 2018). At the landscape scale, using BBS data, tests conducted for arable and mixed farms or landscapes reveal a positive effect of skylark plots on population growth rates (Pringle et al. 2020). No benefit of the plots for invertebrates was found by Smith et al. (2009).

There is some evidence at the farm-scale, but not at population level, that lapwing plots have had a positive association with their target species (Chamberlain et al. 2009); there is also evidence that breeding success is increased on plots under the agri-environment scheme option (Sheldon et al.2007). These were initially compromised by inappropriate placement on farms, because a lack of advice to farmers often led to their positioning too close to woodland or other vertical structures, which the birds tend to avoid. The subsequent delay in the effective implementation of this management option means that the lack of a population response to date may reflect a demographic lag and initial lack of power rather than a failure to address key limiting factors.

Apart from evidence of benefits for birds, MacDonald, Maniakowski et al. (2012) investigated the benefits of fallow plots (in this case intended for the range-restricted stone-curlew (*Burhinus oedicnemus*) in southern England) for brown hares (*Lepus europaeus)*, carabid beetles, vascular plants, butterflies and bumblebees. All groups except carabid beetles were more abundant, more likely to be recorded, or more species rich on the fallow plots than within the crop.

Arable reversion and landscape heterogeneity

Agricultural intensification led to a polarisation in UK agriculture between arable and pastoral systems, with an overall reduction in mixed farming. Individual species and community diversity both benefit from a mixture of crops and grass in close proximity, so this change has been implicated as a cause of biodiversity decline, and measures to reverse it have been included in agri-environment schemes. Arable reversion entails sowing grass on arable land and subsequent management as low-input permanent grassland. (The converse, of converting grassland to arable, has been actively discouraged under the CAP through requirements to restrict the conversion of permanent grassland, although this did not prevent some conversion, for example in SW England.)

There have been few studies of the effects of arable reversion (as an agri-environment scheme option) on birds and the evidence suggests at best a short-term benefit relative to the counterfactual management (Wakeham-Dawson and Aebischer 1998), and there is no clear evidence for large-scale effects on bird population change. Permanent set-aside (former arable land uncropped for 2-7 years) will have had grass cover and was selected over cropped and grass fields by most bird functional groups (Henderson et al. 2000 a,b).

Arable reversion can have significant benefits for insect biodiversity, especially when undertaken to attempt to restore semi-natural grassland. For example, long-term set-aside and arable reversion to species-rich grassland have been shown to positively affect butterfly, moth and bee biodiversity (Alanen et al. 2011; Woodcock, Bullock et al. 2012; Alison et al. 2017). However, benefits to those insect species which are typical of semi-natural habitats are usually contingent on the establishment of the plant species they depend on (Alison et al. 2017). This can be achieved through assisted introductions of those plant species, for example by spreading green hay from nearby semi-natural sites (Woodcock, Westbury et al. 2012). On the other hand, seeding sites with a low diversity grass mix can in fact be detrimental to restoration efforts, as can legacy effects of former management – for example high soil phosphorus (Fagan et al. 2008).

There is evidence, however, that the landscape-scale heterogeneity arising from the interspersion of grassland habitats between arable ones (and vice versa) is positively associated with population trends for a range of bird species (Robinson et al. 2001). The relative scarcity of arable land in Wales probably means that only the converse - pasture conversion to arable - would be supportable for biodiversity conservation reasons. The theory is that the improvement in landscape heterogeneity would have positive effects on biodiversity, but direct evidence is lacking.

5.2 Organic farming

Organic (sometimes called ecological) agriculture can be defined as farming systems where the use of pesticides, herbicides and chemical fertilisers is prohibited (Bengtsson et al. 2005). Organic farms can be arable, livestock, horticulture or mixed systems, and normally operate to government-approved organic standards to assure the status of their produce[13]. Organic systems use crop rotations, natural N fixation, biologically active soil, recycled farm manure (meaning that they are often mixed cropping and livestock systems) and crop residues, and biological or mechanical weed and pest control (Lampkin 1994)to manage crop production. This farming system is considered separately because, although the individual management interventions are clearly relevant to improved land in general, on organic farms the interventions are typically applied and monitored as a package, such that their effects cannot be separated. Therefore, to a great degree, organic management can be considered as a single intervention, which could, in principle, be promoted or supported in a similar way to the other interventions reviewed here.

Evidence of the biodiversity benefits of organic farming is not necessarily clear-cut, particularly in relation to improved land in Wales, because of the different farming

[13] For details of organic standards in Wales, see https://welshorganic.co.uk/join-us/organic-standards

sectors and landscape context of many studies, and the problems of attributing observed biodiversity benefits to specific characteristics of organic farming. Much of the available evidence is focused on the differences between conventional and organic in the arable sector, but some studies/meta analyses do not differentiate between arable and grassland and simply compare organic and conventional (e.g.Bengtsson et al. 2005).

Organic farming was found to promote 30% higher species richness on average and a 50% higher abundance of organisms in a meta-analysis by Bengtsson et al. (2005), but effects differed between taxa and by landscape context, and were largest in studies at the plot scale. A meta-analysis across multiple taxa in arable-dominated systems by Tuck et al. (2014) confirmed that a positive effect is a general pattern: on average, organic farming increased species richness by about 30%, and this differential benefit increases with the proportion arable cover in the landscape, but the size of the effect and the response to intensification varies with taxa and crop type. For example, organic farming is reported to increase diversity of a range of bird prey groups (earthworms, carabid beetles, spiders and aerial invertebrates) on arable land in the Netherlands, but not of all groups in all crop types (Kragten et al. 2011) and diversity of carabid beetles in winter wheat in Italy (Kromp 1989); and diversity of vascular plants in arable field margins in landscapes with differing proportions of organic farmland, in Sweden (Rundlöf et al. 2010).

In a study in Sweden Jonason et al. (2011) found that plant and butterfly species richness was c. 20% higher on organic arable farms than on conventional farms in Sweden, and butterfly abundance was about 60% higher, but there was also a large time-lag effect for butterflies. Butterfly richness responded rapidly, but abundance increased gradually over a 25-year time period after transition to organic. Feber et al*.* (1997) found that the abundance of non-pest species was higher on organic farms although there was no effect on species richness (or on the abundance of the pest species *Pieris brassicae* and *P. rapae)*. Hodgson et al. (2010) found that butterfly densities were higher on organic than conventionally managed croplands and grasslands in a study in 16 landscapes in England.

A range of studies have compared bird communities, and sometimes other taxa in combination, in organic and conventional farming systems. Organic farms typically support higher abundances and more species. For example, Beecher et al. (2002) found that, on average, bird abundance on organic sites was 2.6 times higher than on non-organic sites, and mean species richness per visit was 2.0 times greater. These patterns also held within subsets of species (insectivores, omnivores, granivores and each of three migratory groups), as well as for twelve individual species. No species was more abundant on conventional farms. Further, Fischer et al. (2011) found that organic farming enhanced the species richness of all bird groups during the breeding season, although not in winter, presumably due to the presence of more, and more diverse food resources in the breeding seasons. Ondine et al. (2009) reported that organic farming favoured specialist birds, either considering the whole community or non-farmland birds only. Other studies have shown that farm structure interacts with the specific organic/conventional contrast in management and complicates the patterns that are seen: farm size is also important (more bird species are found on small than large organic and conventional farms, such that small organic farms are the most speciose and large conventional farms the least (Belfrage et al. 2005). Patterns for a positive effect of organic arable management on birds were weak in one study, involving only two of nine in-field-breeding species (Kragten & de Snoo 2008), suggesting that many of the effects that have been reported across the whole organic system reflect farm structure, as opposed to the management of crops.

However, breaking down the components of an organic system can suggest more complex patterns. Chamberlain et al. (2010) analysed bird and habitat data collected on 48 paired organic and conventional farms over two winters to determine the extent to which broad-scale habitat differences between systems could explain overall differences in farmland bird abundance. Density was significantly higher on organic farms for six out of 16 bird species, with none higher on conventional farms, but the organic practices used did not benefit key target bird species that are limited by winter food resources. Patterns of variation in invertebrate food availability between organic and conventional farms are complex: on balance there is greater availability on organic farms, but the pattern varies between crop types (Kragten et al. 2011). McKenzie & Whittingham (2009) suggested that lack of pesticides and increased area of non-cropped habitats on organic farms make a significant positive impact on farmland birds (22% and 15% increases in important bird parameters, respectively), but increased cropping heterogeneity and fertiliser applications on organic farms may both be slightly detrimental to farmland birds.

Several UK studies of bats on paired UK organic and conventional farms found that higher levels of bat activity on organic farms were associated with better habitat structure and quality, particularly of field boundaries such as hedgerows (Pocock & Jennings 2008, Fuller et al. 2005). In the case of three bat species, higher levels of activity were associated with the presence of water features (Wickramasinghe et al. 2003, MacDonald, Morris et al. 2012). This effect on bat activity of habitat structure and quality, rather than of organic management per se, was confirmed in Wales by MacDonald, Morris et al. (2012) who found little difference between Tir Gofal organic farms and their paired counterparts in both habitat structure and bat activity.

Broadening to a wider range of biodiversity, the habitat extent, composition and management on organic farms favours higher levels of biodiversity, so organic farms tend to support higher numbers of species and overall abundance across most taxa, although with a variable effect size; plants showed the greatest responses (Fuller et al. 2005). A review by Hole et al. (2005) found that a wide range of taxa, including birds and mammals, invertebrates and arable flora, benefited from organic management through increases in abundance and/or species richness. However, prohibition/reduced use of chemical pesticides and inorganic fertilisers, sympathetic management of non-cropped habitats and preservation of mixed farming were the key specific aspects of organic farming that benefit wildlife. These can also be implemented in conventional systems, such that the ultimate benefits of organic farming as a system remained unclear, including the value of whole-farm approaches versus targeted, individual, management interventions. Further, positive effects of organic farming on biodiversity do not necessarily translate into clear effects on ecosystem services that biodiversity provides (Macfadyen et al. 2009).

Research has also found that the effects of organic farming are affected by landscape context, and this is clearly important in interpreting the evidence for benefits of organic farming. Several studies concluded that landscape context and heterogeneity may be more important than whether a farm is organic (Bengtsson et al. 2005, Weibull et al. 2000, Weibull et al. 2003, Keulemans et al. 2019), and Bengtsson et al. (2005) found that differences between organic and conventional treatments were more significant at small scale (field) than landscape scale.

Organic farming may benefit farmland biodiversity more in landscapes that have lost a significant part of their former landscape heterogeneity (Smith et al. 2010). This study found a positive relationship between organic farming and bird diversity only in homogeneous landscapes, although the pattern for non-passerines alone was a positive association with organic farming independent of the landscape context. It may be that organic land-use particularly benefits invertebrate-feeders, while seed-eaters are more able also to exploit crop seed resources in homogeneous landscapes and on conventional farms. Rundlöf and Smith (2006) found that organic farming significantly increased butterfly species richness and abundance in homogeneous landscape rather than heterogeneous landscapes, and that organic farming was most common in the latter. In a study in England, organic farms were associated with heterogeneous landscape types, but even in these landscapes they produced greater field and farm complexity than conventional farms (Norton et al. 2009). Ekroos et al. (2008) found that, whilst farms with a larger area of field boundary habitat supported a higher abundance and species richness of butterflies, there was no significant difference between organic and conventional farming.

5.3 Modified management of strips/plots around or within the field

The management of strips or plots around or within the field are considered under the following headings:

- Field margins in general
- Permanent grass buffer strips/margins along field edges or within fields
- Conservation headlands (unsprayed crop); unharvested cereal headlands
- Beetle banks
- Plant nectar flower mixture/wildflower strips
- Plant winter bird food/cover strips
- Leave cultivated, uncropped margins

Field margins in general

Marginal areas in arable fields are less agriculturally productive than field centres due to many factors including weed competition and soil compaction, so they are a good place to take some land out of production (Dickie et al. 2015) to benefit other ecosystem services and also to help to protect hedgerow flora and fauna from pollution and eutrophication as a result of fertiliser and pesticide drift/run-off.

It is well accepted that creating field margins can significantly increase biodiversity, including vulnerable wild pollinator populations (Williams, Ward et al. 2015, Scheper et al. 2013), beneficial invertebrate species (Dickie et al. 2015), plants and birds, although there are multiple variants of the possible management that affects the delivery of these benefits. In a study for Defra (2007), soil invertebrates and earthworms, in particular, were more abundant in the sown margins – indicative of improved soil condition (compared to the cropped land). There was evidence that the complexity and therefore the stability of invertebrate food webs are higher in the non-cropped margins, and particularly those sown with wildflowers.

Different types of margin management deliver different resources or benefits for agri-environment scheme targets, so multi-objective agreements at the farm level would benefit from including multiple types of margin (Vickery et al. 2009).

Permanent grass buffer strips/margins along field edges or within fields

Buffer strips are typically 6m in width and sown between field boundary vegetation and the crop. They have been made available for both arable and pastoral systems in previous agri-environment schemes: in grassland, they typically require fencing to exclude livestock. Wider buffer strips are also used, such as along watercourses or other notable features. Buffer strips both protect boundary features from negative influences such as spray drift and agricultural nutrient run-off, and provide habitat in their own right, contributing to landscape heterogeneity, especially in an arable context. There is strong evidence for benefits of margins to various taxa, albeit with variation in effects with context and specific management after establishment. In general, margins with greater structural and floristic sward diversity will deliver more biodiversity and provide more food resources for birds and mammals (Vickery et al. 2009). Results from one field study in England suggest that creation of grass margins on the edges of arable fields may not adversely impact yield at the field scale, offering a possible win-win scenario for biodiversity and agricultural outcomes (Pywell et al. 2015).

Wolton et al. (2014) found that in autumn and spring, grass margins that were either sown or established through natural regeneration were found to contain predatory beetles (Coleoptera: *Carabidae, Cantharidae, Coccinelidae* and *Staphylinidae*), harvestman (*Opiliones*), spiders (Araneae: *Lycosidae* and *Linyphiidae*) and bugs (Heteroptera: *Anthochoridae*) (Meek et al. 2002). In winter, a diverse range of carabid and staphylinid species and spiders have been found (Pywell et al. 2005), as well as other beneficial species including woodlice (*Isopoda*) and earthworms (*Lumbricidae*) (Smith et al. 2008). Grass margins also support a diverse range of alternative prey including phytophagous invertebrates (Woodcock et al. 2008) and the hosts of parasitic wasps (Powell & Pickett 2003). Marshall et al. (2006) found agri-environment scheme grass margins on arable fields to be associated with high biodiversity of plants, bees and Orthoptera (e.g. grasshoppers) although they did not exceed conventional controls in terms of birds, spiders and ground beetles. Fuentes-Montemayor et al. (2010a) found that micromoth species richness and/or abundance were higher in agri-environment scheme grass margins along field boundaries and watercourses than in conventional controls, although the pattern was less clear for macromoths and only applied in margins along watercourses. Similarly, Merckx et al. (2009) showed that margins have a positive effect on moth diversity and abundance (although a smaller effect than that of hedgerow trees). Alison et al. (2016) found that agri-environment scheme grass margins on arable fields support 1.4x as many grassland generalist moth individuals than conventionally managed margins. Joseffson et al. (2013) also found increased invertebrate activity on fields with grass buffer strips. These fields supported more skylark territories. The researchers were testing the hypothesis that creation of buffer strips would still be of benefit in heterogeneous, complex landscapes and this did seem to be the case. Note, however, that this effect is likely to be context-dependent. In landscapes with hedgerows for field boundaries, these are avoided by species like skylark, so margins will not benefit such species. However, they might benefit hedgerow species that are absent from open landscapes.

Studies of grass margins in pastoral systems (typically fenced off strips along boundaries with varying subsequent management) are less common than those in arable farmland. However, there is good evidence that exclusion of grazing and nutrient inputs lead to increased abundance and richness in nearly all invertebrate groups sampled by Fritch et al. (2017): Araneae, Hemiptera, parasitic Hymenoptera, Coleoptera, Diptera, Isopoda and Dermaptera. However, there were more complex effects of different field margin establishment treatments on the abundance and richness of different taxa, with fencing off the existing vegetation, fencing with rotavation and natural regeneration, and fencing with rotavation and seeding each supporting a distinct invertebrate community. Such variations will affect arable margins as well, suggesting that specific management possibilities should be considered with respect to desired biodiversity consequences.

There is some, although limited and circumstantial, evidence that grass field margin buffer strips benefit common toad, grass snake and common lizard by providing habitat and/or improving connectivity, for example between farmland ponds (Salazar 2014, Salazar et al. 2016).

Preliminary analyses of GMEP field survey data have revealed no selection by birds of agri-environment scheme field margin habitats in pastoral or arable systems, along with other habitats expected to provide summer resources for birds (GMEP BD011). Nevertheless, studies of birds at the field and farm scales have shown a general positive relationship between permanent field margins and bird numbers, such as preferential territory establishment or higher breeding counts in and around margins (Burgess et al. 2015, Davey et al. 2010). Further, floristically enhanced margins appear to be selected for foraging by barn swallows (McHugh et al. 2018). However, such associations have been less clear at the landscape or population level, probably because the management does not address factors limiting population growth. This would be an example of the kind of honeypot effect discussed elsewhere, in which management drives relocation of individuals but not demographic effects that lead to population change. Field margin buffer strips in arable and grass fields under English agri-environment schemes have been investigated separately for their species effects on bird population growth rates, but neither had significant effects on many species that might have been expected to benefit (Baker et al. 2012, Pringle et al. 2020). This may reflect the lack of an effect on key population-limiting factors, i.e. negligible demographic impact.

Lack of active management of field margin habitats could lead to tall dense, species-poor swards (Joseffson et al. 2013), which could have negative effects on ground nesting birds, due to predation risk (inaccessibility forcing nests into vulnerable locations: Morris and Gilroy 2008) or reduced availability of food because the vegetation is impenetrable. The latter problem can be alleviated by mowing part of the margin, such that invertebrates are abundant in the unmown part and accessible to foraging birds when spilling over into the mown part (Douglas et al. 2009).There have been few population-level studies of effects of grass margins on mammals, but there is good evidence from farm-scale research that hedgehogs (Hof and Bright 2010), brown hare (Petrovan et al. 2013) and small mammal communities (voles *Cricetidae*, mice *Muridae* and shrews *Soricidae*; Broughton et al. 2014) use grass margin buffer strips as habitat at least for foraging and/or movement around the landscape. There is also good evidence that small mammal abundance is higher in taller swards in grass margins, i.e. those that are cut only every 2-3 years (Askew et al. 2007). There were no significant effects of grass margins on brown hare population growth rates in an analysis of national survey data (Pringle and

Siriwardena 2017). Further, no effects have been found from several studies on bats (e.g. Fuentes-Montemayor et al. 2011b for common pipistrelle (*Pipistrellus pipistrellus*)).

Further to the evidence for effects of permanent grass margin buffer strips, there is good evidence that more 'enhanced' buffer strips (i.e. with more complex species mixes including forbs that are expected to attract pollinators) add to the benefits provided, at least for invertebrates, because of the floral resources that are then available (Nicholson et al. 2019, Carvell et al. 2007, Marshall et al. 2006, Pywell et al. 2005, Scheper et al. 2015). For example, a study in Germany showed that when creating habitat on arable field margins, increases in hoverfly abundance were greater for flower strips than for grass strips (Haenke et al. 2009). Note, however, that Potts et al. (2009) found that the addition of a wildflower mix to low input grassland had a negative effect on butterfly larval abundance.

Conservation headlands (unsprayed crop); unharvested cereal headlands

Conservation headlands are strips of arable crop along the edge of the field which are not fertilised or sprayed with herbicides or pesticides, and harvested with the main crop or left unharvested. Their design is based on the benefits of sawfly larvae for grey partridge breeding success, and their dependence on arable plants that are killed by herbicides.

Conservation headlands provide considerably more resources for many taxa than either conventional crops or grass margin buffer strips (Vickery et al. 2009). There is good evidence in a study by de Snoo et al. (1998) that the presence and abundance of plants associated with arable farming increase substantially in unsprayed crop margins, and they also found significant increases in flower-visiting insects such as hoverflies (*Syrphidae*), butterflies and natural aphid predators such as ladybirds (*Coccinellidae*).

There is some evidence of a positive effect of headlands on gamebird productivity and breeding numbers at local scales (e.g. Sotherton 1992). However, there is less evidence at larger scales, or for other bird species: few effects on population growth rates were detected from analyses of Breeding Bird Survey data (Baker et al. 2012, Pringle et al. 2020). Uptake has limited the potential to test and to detect effects at larger scales - this management option has not been adopted widely and is small in scale within farms, so is unlikely to benefit large numbers of individual birds.

Beetle banks

Beetle banks are low earth banks across fields, with permanent, usually tussocky, herbaceous vegetation (Wolton et al. 2014); they are designed to provide a source of predatory invertebrates to suppress pests in the adjacent crop. Beetle banks can support diverse invertebrate communities, including cereal crop pest predators (Collins et al. 2002), although evidence for the UK is more limited (Wolton et al. 2014).

In general, beetle banks will rarely be created in more than a very small percentage of fields, so the effect on most bird species, which range over and use much larger areas, is likely to be limited. Grey partridge (with precocial chicks in mobile family groups requiring foliar invertebrates and abundance limited by chick survival (e.g. Aebischer & Ewald 2010) may be the bird species most likely to benefit from this option, but national-level uptake has been too low to allow definitive tests of effects on populations to date. Beetle banks potentially provide nesting cover for partridges, but densities of key invertebrate food groups are lower than those found in grass field margins (Thomas et al. 2001).

Plant nectar flower mixture/wildflower strips

Wildflower strips are sown with pollen and nectar flower seed mixtures which could include legumes, forbs, or fine leaved grasses (Carvell 2007, Defra 2007), for the purpose of providing resources for pollinating and other invertebrates. They may need to be cut and the herbage removed in spring and/or late summer to remove competitive, undesirable species.

Plant species richness in planted wildflower strips increases rapidly although there may be declines over a period of years, meaning that periodic re-sowing is recommended (Defra 2007; Woodcock et al. 2014). The abundance of pollinators has been found to increase with increases in flower numbers (Defra 2007), although there may be variability throughout the year, dependent upon pollen sources. This positive relationship between pollinators and nectar sources has been shown clearly by Pywell et al. (2015) and Baude et al. (2016). Another study found that honeybees did not increase in abundance in an experiment comparing planted and non-planted wildflower strips, however, wild bee and syrphid abundance increased annually in the fields adjacent to wildflower plantings (Blaauw & Isaacs 2014).

Studies across landscapes exhibiting a gradient of agricultural intensity, to investigate the effects of sown flower mixtures and provision of floral resources, showed that sown patches attracted higher densities of worker bees and males and queens (as an indicator of bee reproduction) than unsown controls, the response being strongest in the more intensively farmed landscapes (Carvell et al. 2011, 2015). If sown at a sufficiently large scale there could be spillovers into semi-natural habitats (Carvell et al. 2015). The impact of landscape context is taxon-dependent. In a study by Scheper (2015) increases in solitary bees from implementation of wildflower strips were less likely where there were alternative forage sources, whereas bumblebees increased with increased landscape scale floral diversity. Similarly, Heard et al. (2007) found positive effects of sown forage patches on bumblebee density in England. Positive effects increased in magnitude with the proportion of arable land within 1 km. However, small strips of created habitat can also benefit species associated with semi-natural habitats (Merckx et al. 2010). To maximise benefits to such species, agri-environment scheme interventions may be best placed close to existing patches of semi-natural habitat that contain source populations (Kleijn et al. 2011; Alison et al. 2016).

Of four studies examining the effect on butterflies of agri-environment scheme wildflower margins or pollinator flower mixes (compared to counterfactual controls), three found significant positive effects on adult abundance (Meek et al. 2002; Potts et al. 2009; Korpela et al. 2013) and one found no effect (Brereton 2005). An analysis of English national survey data found that specifically targeted options providing arable nectar resources were positively associated with population growth rates of two butterfly species and negatively with one, but the majority of test results were non-significant (Pringle and Siriwardena 2017).

Studies have found increases in pest predators with wildflower margin strips. Jonsson et al. (2015) found in an experimental study that there was enhanced parasitism of two important crop pests, where they planted floral resource strips (buckwheat). Increases in earthworms and soil invertebrates in the soil margins of planted wildflower strips have been found (Defra 2007, Dickie et al. 2015).

Wildflower strips are not intended to provide resources for birds, but the increased diversity and abundance of a range of invertebrates might be expected to benefit them. Accordingly, corn buntings (*Emberiza calandra*), but not yellowhammers (*Emberiza citrinella*), showed preferential territory establishment in and around nectar flower mix plots (Burgess et al. 2015). However, there is little evidence of effects on taxa relevant to bird diets, and tests of associations with population growth rates at the landscape scale have revealed no clear evidence of impacts. Effects are likely to be limited by the area coverage of the option: few individuals of any species can benefit on a given farm.

Plant winter bird food/cover strips

Research has identified that winter food availability was critical in explaining the long-term declines of granivorous farmland birds (e.g. Siriwardena et al. 2007), and sown cover strips for released gamebirds provided a model system that was proven in respect of farm operations and that has been shown to attract birds in winter (Stoate et al. 2003, Henderson et al. 2004, Parish and Sotherton 2008). Hence, sown strips of seed-bearing, sacrificial crops have been adopted as an option in a range of agri-environment schemes. Crops are typically sown in spring and retained until seed is exhausted and the plants have died down in the following January or February.

In general, there is strong evidence of the association between winter densities of target species and wild bird seed mix (WBSM) crops at the farm scale (e.g. Field et al. 2011, Perkins et al. 2008), and positive effects of WBSM on population growth rates of target species at the landscape scale have also been found in analyses of Breeding Bird Survey data (Baker et al. 2012). Game cover crops may be more attractive to birds in pastoral than arable regions, because of differences in background resource availability (Parish and Sotherton 2008), and the same pattern may apply to WBSM. However, some significant associations are negative for birds (potentially reflecting influences of disease transmission or increased exposure to predation), and unforeseen negative effects of this option may be becoming more common (Pringle et al. 2020). The timing of seed delivery, amongst other things, may be an issue limiting positive option benefits (the peak of demand for seed among farmland birds is in late winter, after WBSM crops are exhausted or have been ploughed in), while net negative effects could show a developing ecological trap, which could arise through the attraction of predators or disease organisms.

WBSM is appropriate for incorporation into arable or pastoral farming systems, but an alternative for pastoral systems is to leave strips or whole fields of intensively managed rye grass to set seed, and then to leave the crop as a winter food resource for birds. Although this has been adopted in Countryside Stewardship in England, it has yet to be evaluated in terms of effects on population levels in practice. However, there is good evidence that it attracts target species to feed (Buckingham and Peach 2005, Peach et al. 2011, Buckingham et al. 2011).

There has been little research into the effects of WBSM on non-bird taxa, but it was one of a suite of agri-environment scheme options that are expected to provide cover for brown hare, and this suite was associated with positive effects on population growth rates in an analysis of national survey data (Pringle and Siriwardena 2017).

Leave cultivated, uncropped margins

Cultivated but uncropped margins have been used in some agri-environment schemes, primarily as an option to benefit arable plants that require broken ground but that are negatively affected by herbicides in cropped areas. However, they do not perform a resource protection function as effectively as permanent grass buffer strips and may be viewed by farmers as a source of weeds, so they have become less popular.

Critchley et al. (2006) investigated the effects of cultivated margin strip treatments on plant communities using a field experiment over three sites on light or chalky soils. The main components of the vegetation were annuals and forbs. The depth and timing of cultivation affected the species composition of the developing flora, suggesting that simply varying the cultivation timing and depth between sites in a given year and over time at the same site would enhance spatial and temporal species diversity. Walker et al. (2007) then showed that plant species diversity, including that of rare species, was higher on cultivated margins than on spring fallow or conservation headlands.

Asteraki et al. (2004) found that naturally regenerating cultivated margins had higher invertebrate richness than most types of sown margin, but that a mixture containing forbs increased plant species richness and maximised heterogeneity of the margin, providing a more diverse habitat for invertebrates. However, soil cultivation causes high mortality of insects such as sawfly and carabids that overwinter as pupae or larvae in the soil and may also affect spiders. While such margins may support a greater abundance and variety of arthropods than conservation headlands, they are usually restricted in suitability to light or shallow soils (Vickery et al. 2009).

In principle, cultivated margins could benefit birds, particularly hedgerow-nesting, ground-foraging species, by promoting food availability and accessibility. Tests of cultivated margin effects on national bird population growth rates have been restricted by the low uptake of the option, meaning that test results may not be reliable and/or representative. Hence, it is not surprising that there were few significant results (5/38 tests conducted), but four of the results were negative (Pringle et al. 2020). Therefore, there is no evidence for benefits of this option for birds at large scales.

5.4 Management of agriculturally unproductive land and features

The management and creation of agriculturally unproductive land and features embedded in improved land are considered under the following headings:

- management of farm ponds
- management of hedges and wooded linear features
- management of farmland trees and woodland
- management of small areas of semi-natural habitats and features embedded within improved land

Note: Evidence for the biodiversity performance of a range of interventions targeted at agriculturally unproductive semi-natural habitats and features is available in **ERAMMP Report-4: *SFS Evidence Review Annex-4 Building Ecosystem Resilience*** and **ERAMMP Report-32: *National Forest in Wales – Evidence Review*** and therefore not covered in more detail here – these include biodiversity management of trees and hedgerows, agroforestry, and the creation of species-rich grassland and different types of woodland on improved land.

Evidence for the contribution of farm ponds, wetlands, trees, hedges and woodlands to Natural Flood Mitigation[14] has been reviewed in **ERAMMP Report-9: *SFS Evidence Review Annex-9 Flood Mitigation*.**

5.4.1 Management of farm ponds

Traditional farmed landscapes included wide distribution of ponds that were maintained for livestock or small-scale irrigation, but these were increasingly neglected during agricultural intensification, leading to accumulation of silt and the development of a closed canopy over the water. The restoration of open ponds is, therefore, an obvious measure by which to restore aquatic biodiversity to farmland. It is also possible that food and other wildlife resources from ponds (apart from water) are exported to 'subsidise' terrestrial biodiversity. There has been only limited research to date into the extent of these 'subsidies', specifically involving birds. Hence, there is some evidence to support benefits to terrestrial species in respect of greatly increased use by most bird species and increased local species richness relative to unrestored ponds (Lewis-Phillips et al. 2019a). Moreover, these effects occurred in multiple seasons, in ways relevant to different aspects of farmland bird ecology (Lewis-Phillips et al. 2019b).

There is considerable evidence that pond creation and restoration benefits amphibians; indeed, any increases to the available stillwater habitat in farmland and the connectedness of dispersed waterbodies is likely to promote amphibian occupancy and abundance (although note that connectivity is also likely to facilitate the spread of disease, one of the principal current threats to common frog *Rana temporaria*: Teacher et al. 2009). A summary of the international evidence for the efficacy of pond management measures for amphibians is provided in Sutherland et al. (2019).

5.4.2 Management of hedges and wooded linear features

Hedges historically provided stock-proofing and definition of ownership boundaries including roads and tracks, while lines of trees and scrub also pick out the sides of watercourses unreachable by livestock. Such features are therefore common in enclosed, lowland farmland and are associated with a range of additional contributions to biodiversity and ecosystem services on improved land. Hedgerows and other wooded linear features on improved land provide a very different range of habitats from the adjacent intensively farmed grassland and arable land, and consequently support a range of plants, invertebrates and birds that might otherwise be rare or absent. Hedgerows other woody features are used by many taxa for foraging, breeding and overwintering, and are also important for habitat connectivity and provide shade and shelter for livestock. The invertebrates that live in hedgerows pollinate crops, control pests in crops, and provide food for bats, birds and mammals. The plants provide alternative pollen sources, food for invertebrates, birds and mammals (berries and seeds), places for lichens to grow and. The soil under them is biodiverse and provides homes for invertebrates (including bees) and mammals. There is good evidence that hedgerows positively affect the richness and abundance of

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[14] Natural Flood Mitigation aims to restore, conserve and enhance natural processes that mitigate flood flows.

flora, invertebrates and birds, and increase landscape connectivity for a set of species broadly described as “edge specialists”.

The diversity of habitats and species associated with a hedgerow tends to be greater where the feature is wider and conditions are more varied. For example, where a zone of less shaded perennial grassland is present, often between the fence and the line of shrubs and trees, then conditions favour an even greater diversity of species with tussock grasses providing nesting habitat for bees, small mammals, a wider range of invertebrates and foraging zones for bats. The combination of shaded and unshaded conditions allows shade-tolerant species to persist but also species of better lit woodland edge habitats. Prospects for increasing the biodiversity of these features are limited by their age and accumulated biodiversity. Attributes that can be managed include width, structural variation, connectivity and the intensity of management of the adjacent improved land. Many species associated with woody linear features often depend on multiple attributes of the feature (Wolton et al. 2013). Although hedges tend to have a negative influence on some species that are associated with open land, such as skylark *Alauda arvensis* and lapwing *Vanellus vanellus*, appropriate management of existing hedges is likely to be neutral for these species.

5.4.3 Management of farmland trees and woodland

In contrast to native woodlands embedded within extensive mosaics of semi-natural habitat in the uplands, farm woodlands on improved land are more likely to be smaller and unmanaged, have high edge to area ratios, and greater exposure to nutrient surpluses and to source populations of weedy, nutrient-loving plant species. Therefore, the same considerations apply as discussed in section 0 when considering reinstating or adjusting woodland management to achieve both positive impacts on biodiversity within the woodland and facilitating connectivity to other linear features and habitats.

In the case of management of farmland trees, the SFS provides an opportunity to avoid the constraints of the CAP Basic Payment Scheme implementation rule that the area occupied by groups of >3 trees less than 10 m apart is ineligible, despite the benefits associated with shade, shelter and biodiversity.

5.4.4 Management of small areas of semi-natural habitats and features embedded in improved land

There are two perspectives for setting biodiversity objectives for improving the habitat condition of agriculturally unproductive habitat patches and/or linear features that are embedded in improved land. The first perspective applies at the individual patch scale where the size and habitat condition of a patch/linear feature *per se* is judged sufficient to justify public funding to improve the provision of associated ecosystem services and/or to enhance the biodiversity value associated with specific habitat types. Collateral benefits to adjacent improved land could arise incidentally or be part of the suite of outcomes expected. The second perspective, focussed on delivering additional (or alternative) farm-scale or landscape-scale benefits for ecosystem services and biodiversity, would require improving the habitat condition of patches /linear features that would be considered too small or in too poor condition to merit funding on their own. An example would be a plan to reconnect and restore fragments of semi-natural grassland, ponds or wetlands coupled with extensifying the intervening improved land matrix. The difference between the two scenarios is that in the latter the size or quality of the agriculturally unproductive features on a farm would make them ineligible for targeted funding to deliver habitat specific benefits in their own right and so the intervention is justified by outcomes defined at a wider scale than the boundary of the patch/linear feature. Examples include areas of grassland too small to graze at a recommended stocking density or a bog too small and hydrologically modified to be considered cost effective to restore. The sub-optimal nature of these remnant features immediately suggests that expectations need to be managed regarding a) the wider spillover benefits that could arise to surrounding improved land from bringing them into management and b) the level of improvement in their condition and extent that might result from their improved ecological connectivity within a matrix that is being managed to become gradually more benign.

The small size and probably poor condition of many agriculturally unproductive habitat patches and features embedded in improved land also implies low resilience, hence they may actually be vulnerable to further degradation when subjected to reinstated management that is intended to have a positive impact. For example, where remnant agriculturally unproductive habitat is extensively grazed as part of a larger mosaic this could result in a net inflow of seeds and nutrients (in dung) and the establishment of weed species from improved land on the unproductive feature. This could be exacerbated by the high edge to area ratio, small size and suboptimal abiotic conditions within the remnant unproductive land (too fertile, too dry etc).

The particular properties of these areas embedded within improved land could therefore lead to very different and, on balance, less positive outcomes when interventions are applied that are otherwise proven to be effective for management of larger areas of semi-natural habitat (see for example Smart et al. (2006) for discussion of the landscape scale potential of unproductive linear features to operate as refugia and recolonisation sources for plants, and how this function changes as surrounding land-use intensifies).

Linear features and small areas of habitat have high edge to area ratio, which means they are more prone to spillover effects from adjacent improved land. This includes exposure to pests, weedy nitrophilous species and non-native species. Moreover drainage, fertiliser drift, disturbance and exposure to enriched run-off can result in conditions unfavourable to species associated with agriculturally unproductive habitats. Small size is also predicted to result in low species richness because of the species-area relationship. Conversely, species richness may be elevated on linear features and small habitat fragments, but these extra species are unlikely to be members of the ‘desirable’ pool if they are weedy generalists that reflect a disturbance gradient across the feature (Smart et al. 2006; Smart et al. 2005). These considerations reduce the value of small areas of agriculturally unproductive land and linear features as donor habitat that can usefully be connected with the improved matrix. They in fact make it more likely that management designed to increase the larger scale dynamism of the system will have a negative impact on agriculturally unproductive habitat remnants unless residual fertility, dispersal and establishment of weeds and nutrient-loving dominants are controlled. Exploiting the donor functions of these unproductive features therefore requires a strategy that carefully balances protection and exploitation. For example, establishing an extensified buffer zone around the unproductive features. This is also consistent with reconnecting patches via managing for a more benign matrix between them.

Place-based targeting will also help. The joint goals of preventing degradation of remnant agriculturally unproductive land and features and managing for reconnection across extensified improved land would be better met by prioritising improved land at the less fertile end of the gradient. Specifically, where the improved matrix was more likely to fall within the *Lolium perenne - Cynosurus cristatus* grassland type (‘MG6’ in the National Vegetation Classification) and therefore effectively semi-improved land. The lower nutrient surpluses associated with such land may also mean that any unproductive features are also less likely to exhibit legacy effects of eutrophication. The detail will differ from place to place emphasising the need for place-based assessment and tailoring of the intervention package as far as is practically possible (e.g. Hayes and Lowther 2014).

Lack of data on the density, size and condition of agriculturally unproductive land and features embedded in improved land in Wales makes it impossible to generalise reliably about their potential to stimulate assembly of resilient networks and contribute to restored ecosystem functioning. The information we have from previous analysis of large-scale surveys certainly argues for managing expectations about the benefits to be gained from bringing small, poor condition unproductive features back into management.

5.5 Other interventions

Two other types of interventions are considered here:

- Provide supplementary food for birds or mammals
- Control predatory mammals and birds (foxes, crows, stoats and weasels)

Provide supplementary food for birds or mammals

Supplementary food attracts birds to feed on it (Conway et al. 2019, Siriwardena et al. 2006) and offers much more control in space and time of food supply to target species than provision of seed via stubbles or sacrificial crops. Trials of feeding resulted in positive effects on population growth rates of target species at the landscape scale, but only when actual bird-use of food patches was considered, as opposed simply to comparison of fed and control areas (Siriwardena et al. 2007). In practice, effective food supplementation depends on the food not being hoarded by dominants, not being exploited by immigrants (who then export any benefits) and not being delivered ineffectively (Wirsing and Murray 2007). Tests of the effects of a real agri-environment scheme option for supplementary food provision have not produced clear results, but this may reflect the fact that the option had only been available for a few years at the time of testing (Pringle et al. 2020).

Outside conservation implementation on UK farmland, supplementary food has been provided successfully for wild hares in North America, but without showing demographic effects, which was interpreted as showing that the population was not limited by food availability (Wirsing and Murray 2007). Feeding is a common component of gamebird management, but often in a package with measures such as predator control and habitat management that make the identification of the effects of individual measures difficult (e.g. Stoate & Szczur 2001). Food is also tightly targeted towards game species, for example via hoppers that restrict access to non-game species. Research on the effects of over-winter feeding on pheasant *Phasianus colchicus* breeding success has concluded that game managers should provide supplementary grain in breeding territories through the spring to increase food availability and to maintain pheasant body condition (Draycott et al. 2005). Supplementary feeding is obviously very common in a garden bird context, but the overall effects beyond attraction of birds to gardens are not known. It seems likely that feeding is having selective effects, because feeding influences almost every aspect of bird ecology, including reproduction, behaviour, demography and distribution (Robb et al. 2008), so effects may be complex, but it is feasible to bring about significant, population-level changes by this mechanism.

Control predatory mammals and birds (foxes, crows, stoats and weasels)

Research identifying predation, particularly of chicks and eggs of ground-nesting birds, as a factor in the population declines (and preventions of recoveries) of priority species, has led to the recommendation of legal predator control (i.e. not including raptors) as a management option. It is important to note that predation may be the immediate cause of mortality or loss, but actually reflect factors such as habitat deterioration or shortage of food leading species to take more risks or to be in poorer condition. Predation being a cause of decline also does not necessarily mean that simply reducing predator numbers will be an effective management strategy: other predators of the same or other species are likely to fill the vacated niche, at least to some extent, for example as a density-dependent response. Further, effects of other environmental influences, such as food availability in winter, may outweigh the effects of predation on populations as the principal limiting factor. Overall, the effectiveness of a predator control activity needs to be tested and shown to be effective in the same way as any other management option.

To date, predator control has not been applied at large scales in a controlled manner and also evaluated in terms of its efficacy in a UK farmland context. Smith et al. (2010) reviewed the evidence for the effects of the removal of multiple predators on multiple responding taxa and concluded that current evidence indicates that predator removal can be an effective strategy for the conservation of vulnerable bird populations. However, this review included island studies, introduced predators and prey, and measures of effect including breeding success as well as population responses at a range of spatial scales. Direct studies of predator control effects have been largely linked to, and confounded with, game management activity (e.g. habitat improvement and supplementary feeding), but have revealed positive effects on brown hares (Reynolds et al. 2010), local bird songbird abundance and breeding success (Stoate and Szczur 2005, White et al. 2014) and moorland-breeding waders (Fletcher et al. 2010). However, a meta-analysis of effects on waders found that the success rate of predator control was highly variable and unlikely to benefit productivity (and therefore abundance) more than expected by chance (Franks et al. 2018). This underlines the importance of the specific predator control measure that is undertaken and the context in which it is applied: this will affect efficacy, and specific evidence of likely benefits is required. There is currently only weak evidence to support an unspecified measure for lethal control of generalist predators.

5.6 Cross-cutting issues for all interventions

5.6.1 Myth busting

Evidence of use is not evidence of benefit

Evidence of use is not evidence of benefit because many elements of biodiversity can be measured in multiple ways, such as counts or presence/absence, winter or summer, plot-scale, field-scale or 1 km square or 10 km square, adult or juvenile (life cycle stage). For different groups, such forms of data vary in ease of collection and quality of evidence provided. However, the latter is not always acknowledged. So, for example, effects of management on habitat selection by mobile species may show redistribution but no change in absolute numbers, while breeding success may change, but be compensated by density-dependence, leading to no net change in population size. In general, effects on certain metrics can be found but these may not influence the factors that limit population size. More specifically, evidence of high counts or densities associated with particular types of management does not necessarily equate to evidence of a population effect.

Understanding what is being measured

Many studies of farmland biodiversity are in terms of species richness. It is important to note that the abundance of particular target species and/or the enhancement of ecosystem services through improvements in biodiversity may well not follow the same pattern. Furthermore, all tests of simple species richness mask the complexity of species turnover as a habitat changes.

5.6.2 Timescale and longevity of impact

Impacts can be delayed because biodiversity management interventions take time to mature. For example, on arable land stubble management or sowing wild bird seed crops should show effects within a year, but with herbicide reduction it could take several years before there is an impact on weed flora.

Critchley (2000) examined the timescale of the effects of non-rotational arable set-aside in a survey of a national sample of 97 sites where management had switched from arable to natural regeneration or sown cover for up to 9 years. In arable dominated regions of Britain vegetation succession remained at an early stage for longer in arable dominated versus mixed farming regions. Hence sites in the west of Britain had greater plant species richness, which the authors attributed to richer landscape-scale species pools. Rapid vegetation change occurred in the first four years as annuals colonised. After this, succession slowed as perennials dispersed and established. Although on some sites species richness was still increasing after 9 years, the study showed that species-poor communities were likely to result in the absence of direct introduction of propagules. This constraint was also more likely to be operate in landscapes with smaller and less diverse species pools. Species richness in general declined with distance from field boundaries indicating that these linear habitats are an important recruitment source. This gradient also indicated that dispersal and establishment was more likely in closer proximity to linear features even after 9 years of succession. The study concluded that the species composition of stable spontaneous ex-arable grassland assemblages on set-aside will be determined by local site conditions and management, and be dependent on soil properties and proximity to propagule sources.

Restoration of diverse floral habitats adjacent to high-value pollinator-dependent crops can increase pollination and pay for habitat installation in three to four years (Blaauw & Isaacs 2014), and planted strips will show results in the first year. However, there is some uncertainty over the longevity of the beneficial impacts of pollinator strips based on the species sown, seed provenance and whether strips have a permanent location within fields (Dickie et al. 2015). For example, longevity may be limited if legumes form a diminishing proportion of the sward over time (Woodcock et al. 2014).

For mobile species, many interventions operate on two timescales: more-or-less immediate for use of the option and longer-term for demonstrable population changes. Otherwise delays can occur because target species take time to find newly created or newly managed habitat, or because effects are cumulative over time and a signal of population increase needs to reach a certain size before it is detectable through sampling noise. Further, particular management may only provide benefits in bad years (when ambient resources are short and need to be augmented) or good years (when numbers are free to increase in the absence of other constraints). Hence, an overall benefit would only be detectable after a sufficient run of years to encompass such good or bad years.

5.6.3 Magnitude of impact

Some key studies have detected significant effects of key agri-environment scheme options, but with small effect sizes. This implies that the management is having an effect, but one that is small relative to other influences, such as causes of population decline: benefits can be statistically significant without being sufficient to meet conservation aims.

A related issue is that management can have effects that are detectable as local responses, but not necessarily at the population level. This could be because the resources provided by the management are not limiting for populations (i.e. there are bottlenecks in respect of other elements of the life cycle, such as late-winter food for birds (Siriwardena et al. 2008) or early-season floral resources or nesting habitats for bees (Dicks et al. 2015), or because benefits are exported (e.g. biological productivity is increased but new individuals are recruited outside the farm boundary, or winter management increases survival but the individuals affected breed elsewhere). It may not be possible to identify when relationships like these are occurring in monitoring data, so an element of expert judgement based on ecological knowledge about the species concerned is valuable, for example to interpret whether a given option and its rate of uptake is likely to deliver sufficient resources to address the limiting factor for abundance.

More broadly, the philosophy underlying management action could be to aim to deliver national population responses (which could be compromised by the influence of external factors such as climate change or conditions on wintering grounds), or to benefit target species as much as possible within the limits of their ecology within the focal territory. For example, management could seek to reverse a decline in a migratory population, or just to maximise use of local habitat by the species. The latter might not deliver a population effect but could be valuable in doing what is possible to help a priority species within the context of the portion of the life cycle that occurs in Wales. Management that has only local benefits might therefore still be justifiable, but expectations of the effects that it will have in practice need to be proportionate. This suggests that the application of management to species with complex life cycles, e.g. using multiple habitats or migrating, would benefit from specific ecological advice with regard to the setting of targets.

The first agri-environment scheme in Wales to be evaluated in detail was Tir Gofal. Two studies have evaluated this scheme, considering all relevant options in agreements, which were dominated by management of improved land. First, Macdonald et al. (2019) conducted ‘snapshot’ surveys of multiple taxa at the farm and field scales, as well as incorporating analyses of some existing data sets for priority species. They found that there were few differences between Tir Gofal and control locations, with the exceptions involving arable specialists; brown hare, yellowhammer and arable plants. This study may have been limited by the lack of an explicit temporal dimension, i.e. comparing changes between areas with different management regimes. Dadam & Siriwardena (2019), as a part of GMEP, analysed long-term, historical bird survey data for Wales and found similarly that arable management options were the most successful, along with the management of woodland and hedgerow (see Evidence Review 4, section 4.5). Even in these cases, effect sizes were small. Moreover, the lack of positive effects for grassland management in both studies is notable given the predominant land-uses in Wales. This does not imply that management of grassland *per se* is less valuable than that of arable land, however. In fact, there have been more and more creative management options developed for arable systems than for grassland, and these options may involve more significant changes to land-use and the habitats that are available to target species. Further, while there are exceptions to this, such as sacrificial seeding rye grass for farmland birds (Buckingham et al. 2011), these are recent additions to the agri-environment scheme portfolio that do not have high uptake and sufficient time may not have elapsed to allow the demonstration of large-scale effects. Nevertheless, further development or evaluation of grassland-specific measures may be warranted.

5.6.4 Spatial context and connectivity dependence

This is discussed in Section 6 below.

5.6.5 Metrics and verification

For verification and payment control purposes, infield grassland interventions and infield arable interventions involving input reductions, and mowing, grazing or tillage interventions would currently require measurement in the field. For most of the other interventions reviewed, including field/crop margins, ponds and woody features their presence could be identified remotely, at different times of the year, but measurement in the field would still be needed to determine habitat condition, including metrics related to habitat function and connectivity.

Measurement of plant diversity and abundance (surveyed in quadrats, 2m x 2m) would be suggested for field margins and linear strips along arable field margins (1m x 100m). However, the size of the unit recorded needs to be scaled to the size of the organism, and whether there is interest in capturing turnover of rare as well as common species. For example, nested quadrats can be used with a larger total area recorded in woodlands or in species poor habitats where the initial stages of colonisation (few individuals and low abundance) need to be detected. Plants surveyed may also be those defined as agricultural weeds.

In the case of invertebrates, the abundance and species richness of pests, pest predators and pollinators can be measured using pitfalls, soil cores, sweep netting, transect walks or pan traps (Jonsson et al. 2015, Carvell et al. 2015).

Verification for impacts on soils greenhouse gas emissions, water quality and flow etc have been covered in previous reviews.

5.6.6 Co-benefits and trade-offs

Agricultural productivity and displacement

Without any inputs of fertilisers, herbicides or pesticides on improved land there are likely to be lower yields and crop losses from disease or damage. However, there is limited quantitative evidence on the impacts of reducing pesticides and herbicides on yield quantity and quality (Keulemans et al. 2019), and the impact will be affected by the level of usage before the reduction is applied. Where application of pesticides and herbicides is high there may be potential for reduction with no adverse effect on yield. In France, a study of arable farms suggested that low use would not reduce high productivity or high profitability of arable crops in 77% of the farms (Lechenet et al. 2014). Jacquet et al. (2011) suggested that a reduction of plant protection products in French field crops is possible by 30% without reducing farmer's income. It is well accepted that in field margins, by reducing disturbances and providing resources for pest predators (such as carabid and staphylinid beetles), natural attack rates on pests can be increased so reducing their impact (Jonsson et al. 2015, Landis, Wratten and Gurr 2000). This can lead to reduced pesticide use (Firbank et al. 2011). More diverse crop rotations can mitigate the impacts of failing/poor performing crops and improve yield stability (Defra 2018).

In productive agricultural systems in lowland England, positive effects of organic farming for plants, bees, butterflies and other arthropods are linked to roughly proportionate reductions in yield (Gabriel et al. 2013). Yield reductions will be dependent upon crop type, reductions in yield of between 20-40% (19% for wheat and 42% for potato) have been reported (Keulemans et al. 2019). Seufert (2019) estimated 25% as an average yield loss for organic systems. These yield losses are context dependent (soil type, crop type, management) and also have been found to reduce with time since conversion (Seufert 2019). Applying other cropping diversification practices (such as multi-cropping and crop rotations) can reduce the yield gap (Ponisio et al. 2015).

No-till on its own reduces yields. When it is combined with residue retention (cover crops) and crop rotation, its negative impacts are minimized (Pittelkow et al. 2015). There are also possibilities of crop failure in reduced tillage systems (Freibauer et al. 2004).

The trade-off between agricultural intensity and plant biodiversity across European grasslands has been well quantified (Kleijn et al. 2009); plant biodiversity declines with land-use intensity, declining most steeply on extensive grasslands. Nonetheless, some interventions, for example grass margins on arable fields, may benefit biodiversity without compromising yield (Pywell et al. 2015), demonstrating how conservation on farmland does not always come at a cost to agricultural productivity.

ERAMMP Report-2: *SFS Evidence Review Annex-2 Sward Management* discusses productivity-diversity relationship in improved grasslands. It is not just the yield of the grassland that matters, but also the quality of the sward. Shellswell (2017) concludes that the metabolisable energy and crude protein of most species-rich grasslands is not equivalent to that of agriculturally improved grasslands, citing figures from a number of sources. However, there is other evidence that diverse swards can be more productive, e.g. one study found that diverse swards resulted in an average 12% increase of live weight gains (across grazing systems and livestock species) (Jerrentrup et al. 2020). There is limited evidence that the herbal component of a grassland/ley has higher quantities of minerals/macronutrients than grass and legume sward components (Lindstrom et al. 2012, Pirhofer-Walzl et al. 2011, García-Ciudad et al. 1997). The use of species-rich grasslands within an agricultural system may reduce the need for mineral licks to maintain healthy livestock (Shellswell 2017). There is limited evidence that species rich grasslands provide opportunities for livestock to use certain plant species that contain compounds to help prevent or reduce illness (Shellswell 2017).

If management interventions cause a reduction in yield then either yield reductions will need to be accepted (possibly in association with measures to reduce food waste) or more land will be required to grow crops which may involve conversion of semi-natural habitat elsewhere (Keulemans et al. 2019).

Water quality and flood mitigation

See also ERAMMP Report-1: *SFS Evidence Review Annex-1 Soil Nutrient Management for Improved Land* and ERAMMP Report-9: *SFS Evidence Review Annex-3 Flood Mitigation*

Potentially there may be a reduction in nitrate leaching from the soil from crop rotation, resulting from the varying ability of crops to remove nitrate from different places in the soil profile, resulting from their different rooting depths and densities. Manure application can be associated with nitrate leaching (Moxley et al. 2014), especially where application is excessive or poorly timed (Goulding et al. 2000, Powlson et al. 2011). In soils already high in phosphorus, addition of composts and manures carries with it a risk of phosphorus runoff. In a review of phosphorus management of organic manures, Smith et al. (1998*)* concluded that restricting topsoil extractable phosphorus levels to 70 mg l^{-1} should minimise the risks of unnecessary phosphorus enrichment and subsequent leaching. Increased herbicide usage may be required under reduced tillage unless used with other options, which can increase the risk of groundwater contamination and soil erosion, if leaching occurs, and may have adverse effects on human health (Carmona et al. 2015, Alleto et al. 2010; Gasnier et al. 2009). Cover crops, particularly legumes, and improvements in sward diversity, can increase soil mineral nitrogen availability, allowing the reduction in use of chemical nitrogen fertilisers and lower the risk of nutrient leaching (Eory et al. 2015). However, if soil is left bare after a legume harvest, there is a risk of residual nitrogen being lost by leaching (Defra 2018). Field margins can act as buffer strips to intercept and retain sediments (Dickie et al. 2015).

Many of the management interventions on improved land reviewed above (e.g. reduced tillage, improved sward diversity, use of organic fertilisers, increases in the amount of semi-natural land including hedgerows and field margins) can be associated with improvements in soil structure, reduced soil compaction and increased rates of water filtration that can improve water retention and reduce the risks of flooding (Freibauer et al. 2004, Borin et al. 2010, Carroll et al. 2004, Maskell et al. 2019). Infiltration may also be improved by deep-rooting species such as legumes. Cover crops require water, however the presence of a permanent mulch of crop residue reduces surface runoff (Sun et al. 2015). It also reduces the amount of solar radiation reaching the surface and evaporating water, and soil moisture is more likely to be retained under cover crops (Scopel et al. 2004).

Air quality

See ERAMMP Report 8: *SFS Evidence Review Annex 8 Improving air quality and well-being*.

Green-House Gas (GHG) balance

See also ERAMMP Report 3: *SFS Evidence Review Annex 3 Soil Carbon Management* and ERAMMP Report 7: *SFS Evidence Review Annex 7 Systems Approach to GHG emissions reduction*

Co-benefits of reduced tillage include reduced costs and GHG emissions associated with fuel consumption (Buckingham et al. 2013, Govaerts et al. 2009). There may be positive effects on soil moisture (Freibauer et al. 2004) and soil structure (Powlson et al. 2014). Trade-offs of reduced tillage include a risk of increased N_2O emissions in poorly aerated soils (Freibauer et al. 2004, Rochette 2008). The GHG mitigation potential of reduced tillage could be reduced by 50-60% after consideration of increased N_2O emissions (Freibauer et al. 2004).There is conflicting evidence about the impact of reduced tillage on soil organic carbon (Angers and Eriksen-Hamel 2008, Baker et al. 2007), particularly the distribution at depth (Luo et al. 2010, Dimassi et al. 2014). A review concluded that reduced tillage is not a reliable option to increase the soil organic carbon of UK soils (Buckingham et al. 2013) or globally (Luo et al. 2010).

While there would be clear benefits for climate change mitigation due to the carbon footprints associated with input reduction, N_2O emissions are a possible trade-off from manure, sewage sludge and urban compost, although emissions from production of inorganic fertilisers could be reduced by using organic fertilisers (Freibauer et al. 2004). Jones et al. (2006) found that for some types of organic fertiliser increases in soil organic carbon were outweighed by N_2O emissions, given the increased global warming potential of N_2O. However, this offset may be preventable by using appropriate manure application techniques (Misselbrook et al. 2002) and modified manure like acidified slurries (Fangueiro et al. 2015). Anaerobic digestate may be a further means of reducing GHGs to help to meet carbon budgets, but there may be a significant constraint in the availability and location of sufficient land for spreading digestate in a sustainable way, especially where this is produced off-farm.

Soil quality

Crop rotations and cover crops and use of organic inputs can have strong beneficial effect on soil quality and structure (Schipanski et al. 2017, Defra 2018, Freibauer et al. 2004). Using crops with different root architecture and rooting depth allows access to immobile macro and micronutrients in different parts of the soil profile (Defra 2018). Cover crops can be used to eliminate bare ground, potentially increasing soil nutrients and soil organic carbon by increasing productivity (especially if plant residues are returned to the soil), preventing erosion (Buckingham et al. 2013, Desjardins et al. 2005), improving soil structure through rooting systems (Scopel et al. 2013), or altering the soil bacterial diversity and composition to improve nutrient cycling (Alahmad 2018. Manure can also be beneficial for the soil microbial community (Kallenbach and Grandy 2011).

Functioning semi-natural habitats

See ERAMMP Report 4: *SFS Evidence Review Annex 4 Building Ecosystem Resilience.* Improved land, especially some semi-improved grasslands, could have a significant role in semi-natural habitat restoration and creation, which is an essential component of landscape-scale management of semi-natural habitats in Wales.

6 Significance of context, synergy and scale of biodiversity interventions on improved land

6.1 Effect of context on the potential benefits of interventions on improved land

The biodiversity effectiveness of almost all the interventions reviewed here is influenced, to differing extents, by the choice of location relative to existing habitats and to bio-physical conditions at field/farm level, and by the scale (patch size) of the intervention. For example, Scheper et al. (2013) carried out a meta-analysis of agri-environment scheme effects on pollinators across Europe, finding greatest positive impacts in croplands (as opposed to grasslands) in landscapes with small amounts (1-20%) of existing semi-natural habitat. Such considerations are particularly important for design of the SFS because evidence from earlier agri-environment and set-aside schemes has shown that, in practice, individual farmers' decisions on the type, location and scale of the interventions on improved land tend to be strongly influenced by the objective of minimising the potential negative impact on the farm business.

6.2 Synergy of interventions on improved land

Another important factor is the added biodiversity benefit of implementing several different interventions as a package at farm or field level – the effect of the whole is greater than the sum of the parts. Here we consider the added benefits of synergistic implementation of complementary interventions on improved land, where relevant.

For example, on arable land reduced tillage, crop rotation, organic inputs and cover crops are often implemented together. Lack of soil inversion increases weed infestation (and requires more herbicide application) under reduced tillage, while crop rotation and cover crops can assist with weed management. They can also help to ameliorate yield losses under reduced tillage. Application of organic matter and reduced tillage may increase nitrogen leaching and N_2O emissions, but cover crops can reduce nitrogen leaching. There is conflicting evidence about the impact of reduced tillage on soil organic carbon, but cover crops and application of organic material can increase soil organic carbon. Reductions in fertilisers and pesticides can be implemented with reduced tillage, cover cropping and crop rotation, practices which add plant nutrients and help to increase beneficial arthropods and to control pests so fewer inputs are required.

In the case of birds, the hypothesis that combining management interventions that benefit, say, nesting and feeding, should lead to synergistic effects on individual species was tested directly on birds in England, but with little evidence of clear effects (Pringle et al. 2020). It is likely that numbers on a given farm are limited by just one factor at a time, so combining options at this scale has little benefit. However, at the landscape scale, multiple option types can have positive effects on the same species, which may reflect differences in the limiting factors that are operating in different places. However. there are potentially further synergistic (and antagonistic) effects of individual measures at the community level. Clearly, interventions targeted at different species could promote the local populations of both of them, albeit with trade-offs between the quantities or areas of management that are devoted to each. This concept, also including the potential for adjacent, contrasting land-uses to provide complementary resources required by species, underlies proposals to promote farmland biodiversity by increasing heterogeneity (Fahrig et al. 2011, Sirami et al. 2019). Conversely, management to promote, for example, open field species, may be compromised by the presence of management promoting taller hedgerows. Further, interventions may be ineffective if implemented without sufficient consideration of landscape context, such as aiming to promote open field species in a highly wooded landscape (e.g. Chamberlain et al. 2010).

Synergy with farm and catchment scale nutrient control measures

Evidence exists detailing the efficacy of interventions that reduce nutrient surpluses on agricultural land. Collins et al. (2018) carried out a model and stakeholder assessment of the cost-effectiveness of a range of nutrient control measures based on agricultural land in England which identified the highest ranked measures for reducing on-farm agricultural diffuse pollution as follows:

1. Use a fertiliser recommendation system.
2. Do not apply manufactured phosphorus fertilisers to high phosphorus index soils (those with an Olsen soil phosphorus index of 4 or above).
3. Move feeders at regular intervals.
4. Leave over winter-stubbles.
5. Do not apply manufactured fertiliser to high-risk areas.
6. Site solid manure heaps away from watercourses/field drains.
7. No overgrazing of natural or semi-natural grassland.
8. Do not spread slurry or poultry manure at high-risk times.
9. Do not apply manure to high-risk areas.
10. Increase the capacity of farm slurry stores to improve timing of slurry applications.
11. Incorporate manure into the soil.
12. Avoid spreading manufactured fertiliser to fields at high-risk times.

Withers et al. (2017) recommended that mandatory soil testing be enforced in catchments suffering from or susceptible to eutrophication. Frequent soil testing would also make the use of fertiliser planning tools more effective. However, failure to act on soil testing results can arise as a result of complex cultural or embedded perceptions about the cost-benefits involved (Prysor Williams pers.comm; Gibbons et al. 2014).

Recent research has achieved reasonable success in establishing which measures work, identifying those most acceptable to industry and stakeholders on grounds of cost and practicality (Collins et al. 2018) and synthesising this evidence into detailed practical guidance for land managers (Cuttle et al. 2016). Because of the focus on achieving regulatory targets linked to water quality the evidence is rarely pitched at the measurement of terrestrial biodiversity targets or ecosystem resilience. However, model products such as Farmscoper or Ncycle (e.g. Gooday et al. 2014, Scholefield et al. 1991) would potentially allow prediction of the effects of management on other targets of pollutant management.

The relevance of these interventions to restoration of biodiversity and ecosystem resilience on improved land can be judged in two respects. First, by assessing how likely each intervention is to impact in-field fertility and then by adopting a landscape-scale perspective, in which interventions would be rated more highly where they are effective in reducing exposure of other habitats, wetlands, watercourses and green infrastructure to nutrient surpluses arising from the management of improved land in which they are embedded.

Barry and Foy (2016) examined the chemical and biological effects of control measures on 40 streams in two catchments in Northern Ireland between 1990 and 2014. Both areas were dominated by cattle production and, during this period, saw implementation of a number of policies and associated nutrient reduction measures. Diminished pollution by organic wastes[15] meant that 85% of streams exhibited chemistry suitable for salmonids in 2009 compared to 40% in 1990.

The recovery period was associated with a range of policy levers including implementation of the Nitrates Directive, CAP-driven decoupling of livestock numbers from farm direct payments, with cross-compliance as a condition of support. Associated interventions included a) minimum spreading distance from watercourses, b) mineral fertilisers containing phosphorus permitted only if a valid soil analysis showed a requirement, c) financial penalties for non-compliance and d) closed seasons for application of mineral fertilisers and livestock wastes. While partitioning of the variation in the ecological effect to specific measures was not and could not be achieved given their co-location, grant aid for construction of slurry stores was highlighted as especially useful in giving farmers the flexibility to plan slurry spreading enabling them to reduce application and applying wastes in a more targeted manner.

The appreciable and significant declines in phosphorus and nitrogen in watercourses over the period suggests that all or some combination of these landscape-scale interventions are effective in reducing nutrient loads to improved land and embedded wetlands, green infrastructure and semi-natural habitats. (A caveat here would be that if buffer strips along watercourse banksides are effective in intercepting enriched run-off, then any residual or managed reinstatement of biodiversity on these features may be subject to eutrophication, despite the benefits for the adjacent watercourse.)

6.3 Landscape-scale planning of SFS management interventions

As ERAMMP Report 4: *SFS Evidence Review Annex 4 Building Ecosystem Resilience* noted "*it is important to recognise that all habitat creation and restoration actually happens at the individual farm scale and the cumulative impact of this underpins the extent and connectivity of habitats which is a key contributor to ecosystem resilience. Effective improvement in habitat connectivity requires planning and targeting interventions spatially at the landscape scale and implementing them 'in the right place' on many individual farms. This approach requires management action on both semi-natural habitats and improved land, with significant opportunities to enhance ecological resilience through habitat creation as well as through improving the condition of existing habitats*." Position of interventions in the landscape is equally important for supporting ecosystem services, and there may be trade-offs or synergies in terms of optimal location depending on the potential environmental benefits and priorities. Moreover, a landscape-scale and perhaps even

[15] During this period manure production in the study catchments declined by 7%, while regional chemical fertiliser inputs declined by 37% for nitrogen and 79% for phosphorus, and the regional nutrient surplus was lowered by 18% for nitrogen and 49% for phosphorus.

larger scale perspective is required to manage for consequential secondary effects; for example, this might arise where extensification or habitat restoration interventions are compensated by intensification of improved land elsewhere.

In the case of improved farmland the differences between the habitat and the surrounding matrix are greater so the composition and configuration of semi-natural features is important, and corridors are more significant because the permeability of the matrix is reduced (Concepción et al. 2012). Some authors have predicted and found that the effects of agri-environment measures on species richness will be maximal in landscapes of intermediate complexity (Tscharntke et al. 2005), decreasing to zero in the simplest and the most complex landscapes (Concepcion et al. 2008, Concepción et al. 2012). The assumption is that relationships between landscape complexity and field scale species richness are non-linear, and that these interact with local management effects (Concepción et al. 2008, Concepción et al. 2012).It is argued that species richness in agricultural fields increases from simple to complex landscapes because semi-natural landscapes contain more diverse resources and dispersal corridors (Concepción et al. 2012, Benton et al. 2003). In simple landscapes there are few available species to colonise newly created habitat. Therefore, as landscapes become more complex, local diversity is expected to increase until a saturation point is reached from where no further increases in species richness are expected. This saturation may be due to continuous recolonization filling all available niche space or to potential negative effects of semi-natural habitats on species requiring more open conditions (e.g. farmland birds) (Concepción et al. 2012, Maskell et al. 2019). However this hypothesis is not necessarily demonstrated in all landscapes.

There is good evidence that the performance of agri-environment scheme measures on farmland does vary with the landscape context. Building on a body of previous research (e.g. Concepción et al. 2008), Batary et al. (2011) conducted a meta-analysis of studies of both species richness and abundance across multiple taxonomic groups. In arable land, agri-environment schemes significantly enhanced species richness but not abundance, in simple landscapes(<20% semi-natural habitat nearby) but not in complex landscapes (>20% semi-natural habitat nearby). In grassland, agri-environment schemes effectively enhanced both species richness and abundance regardless of landscape context. For example, pollinators were significantly enhanced by agri-environment measures in simple but not in complex landscapes, in both arable and grass. Similarly, Scheper et al. (2013) carried out a meta-analysis of agri-environment scheme effects on pollinators across Europe, finding greatest positive impacts in croplands (as opposed to grasslands) in landscapes with small amounts (1-20%) of existing semi-natural habitat. Thus, targeting interventions toward simple landscapes could maximise overall increases in species-richness and abundance and, in the case of key functional groups such as pollinators, could also increase ecosystem service provision. However, when the aim is to benefit specialised species associated with semi-natural habitats, it may instead be best to target interventions toward complex landscapes where source populations exist (Kleijn et al. 2011, Alison et al. 2016). These results suggest that targeting and implementation of SFS biodiversity interventions should be adapted with respect to landscape structure and target species groups.

A further consideration is the interaction of the effects of multiple adjacent agri-environment scheme agreements on different farms: either because individuals of mobile species can readily move across farm boundaries or because amalgamated habitat quality across a local area allows populations to build to a sustainable size that cannot be maintained on a single farm. Hence, field-level bird responses to agri-environment scheme have been found to be higher where more of the surrounding landscape is included in an agri-environment scheme (Dallimer et al. 2010) and it is common for population growth rate responses to management to be stronger at a spatial scale that is considerably larger than the scale of biodiversity sampling (Baker et al. 2012).

It should be noted that effects in many studies are in terms of species richness; abundance of particular target species or enhancement of ecosystem service delivery via improvements in biodiversity may well not follow the same pattern. Moreover, all tests of simple species richness mask the complexity of species turnover, such as the loss of the few species that are associated with simple landscapes in the transition to complex ones.

This is particularly important in the case of interventions which support the management or creation of feeding, breeding and shelter for mobile species associated with improved farmland landscape (birds, amphibians, pollinators and other invertebrates). For example, Geiger et al. (2010) did not find a positive relationship between organic farming and bird species diversity which they suggested could be due to the large spatial scale of the pollution associated with pesticide use across Europe, which inevitably leads to negative effects of pesticides – even in areas where the application of these substances has been reduced or terminated, and this applies particularly to species operating at larger spatial scales such as birds. Note, however, that many benefits associated with organic farming are likely to be due to habitat effects, such as the need for mixed farming leading to (a) higher heterogeneity of land-use, (b) smaller fields and (c) more developed hedgerows. It is debatable whether these changes are considered to be intrinsic to organic management, but they complicate the ascribing of changes to the absence of chemical inputs, e.g. on hedgerows and other patches of non-cropped ground that are important in agricultural landscapes for the existence of healthy and diverse pollinator populations (Wolton et al. 2014, Haenke et al. 2014, Morandin & Kremen 2013). They can also enhance populations of the natural enemies (predators and parasites) of crop pests by providing a wide range of microhabitats across the shrub layer, trees, banks, base, margins, ditches and soil, nectar and pollen resources, and larval development resources. Jonsson et al. (2015) found that experimental provision of floral resources enhanced parasitism rates of two globally important crop pests in moderately simple landscapes but not in highly complex ones, and this translated into reduced pest abundances and increased crop yield. However, a wide range of microhabitats in complex landscapes may also benefit pest species. Critchley et al. (2003) also highlighted widespread lack of vegetation response to restoration management in UK Environmentally Sensitive Areas, highlighting the role of residual fertility and lack of propagule sources. This was especially apparent in improved land. In order to maximise the chances of successful restoration and achieve spatial targeting of interventions Török et al. (2018) recommended an assessment of landscape-scale dispersal sources, for example identifying the density of residual semi-natural habitats and the density of linear features. They also recommended the creation of small-scale heterogeneity but also highlighted the need to control establishment of undesirable species that are also likely to benefit from interventions favouring dispersal across landscapes.

7 Evidence Gaps

Notable gaps include:

1. Evidence to identify best practice for incorporating small areas of agriculturally unproductive land and features into farm and landscape-scale management plans for improved land, including semi-improved grasslands: and how the particular properties of these individual patches, such as size, habitat condition and management intensity of the surrounding land, will affect the outcomes of applying habitat management measures proven to work elsewhere.
2. Evidence to identify best practice for optimising the placement and management of new and existing wooded linear features in improved land to achieve multiple benefits for ecosystem resilience.
3. Evidence has been presented on the importance of landscape context in the location and targeting of agri-environmental measures, however, results have been found to be inconclusive. They vary by farm type, intervention type, environmental outcome and by taxa. More landscape level research, relevant to Wales, is required.
4. Evidence for impacts on different target groups of interactions between interventions (on the same land and on nearby land) at both field and landscape scale,
5. Evidence of costs of implementation in different contexts, and of trade-offs and synergies with agricultural production and other SFS objectives (e.g. soil carbon, nutrient management and flood mitigation).
6. Evidence is lacking about how realising long-term, large-scale objectives for extensifying improved land are likely to be constrained or deviate from expectation because of the legacy effects of intensive land-use. For example, what are the most cost-effective solutions for overcoming the 'unhelpful resilience' (sensu Standish et al. 2014) of improved land.
7. Guidance on timescales required to achieve reassembly of resilient networks given different legacy levels of improvement.
8. Practical advice on how to measure resilience and realising the synergies required among the components of resilience (diversity, extent, condition, connectivity, adaptive capacity) that can show when resilience has been or is being achieved.
9. A general evidence gap is that quantitative evidence for how much of a given intervention is needed to deliver a given benefit is almost always lacking, and is also likely to be context-specific. Therefore, we can identify interventions that provide qualitative benefits, but precise information about whether more or less of it is required to meet a policy target needs monitoring and evaluation of the management once it is in situ, probably followed by modelling to predict wider consequences. For example evidence is lacking on reductions in pesticides. Much of the current evidence comes from organic farming where there are no pesticides and a suite of other measures which, together with increased habitat heterogeneity, can complicate interpretation.
10. There is an imbalance between work done in improved grassland and arable land (interventions in the latter have seen more research, development and

evaluation). In Wales there is the opposite imbalance in land-use, and therefore in more widely relevant management,. This is not so much an evidence gap as an evidence bias: general conclusions about agri-environment scheme impacts based on the range of current evidence could be misleading for the Welsh context, so the general relevance to Wales of evidence for intervention effects should be considered in making high-level decisions. This includes evidence around organic systems, which have been studied extensively and are relevant to Wales, but have (to date) not been implemented widely.

Filling these evidence gaps requires long-term, large-scale measurements in which place-based case studies can be embedded. In this respect Wales ought to be well-served by the data and opportunities arising from the ERAMMP programme (https://erammp.wales). This monitoring and surveillance program is designed to be responsive to policy driven needs for evidence at multiple scales. For example providing unbiased and representative measurement of ecological change from farm to field to landscape but also providing the wider context against which the generality of local place-based evidence gathering can be better understood.

It is strongly recommended that an SFS evaluation and monitoring system is designed and rolled out in parallel with the SFS. This should be of robust design, capable of evaluating the effects of interventions (singly and in combination, at different scales) e.g. before-after control-treatment, and taking account of additionality and 'deadweight' (effects which would have occurred anyway in the absence of intervention).

8 Summary

This review covers evidence for management to improve biodiversity related features of improved farmland, and should be read in conjunction with two other ERAMMP Evidence Review Reports, The first is ERAMMP Report-4: *SFS Evidence Review Annex-4 Building Ecosystem Resilience*[16], which covered semi-natural habitats within farmland and common grazings (including hay meadows, calcareous grasslands, farm woodland, hedgerows and trees within farmland) but specifically excluded improved farmland. The second is *the National Forest in Wales - Evidence Review* Reports 32-37[17] which covered biodiversity improvements to farm woodlands including creation/expansion, of broadleaf woodland/agroforestry, connectivity and small woody features, and management of under-managed woodlands.

The links between these reviews is particularly important for the design of the SFS because:

- in Wales the distinction between agriculturally improved land and semi-natural habitats is not clear cut, and there are significant areas of semi-improved permanent grassland habitats, often in mosaics with improved grassland or other semi-natural habitats some of which retain potential for habitat improvement. In the SFS the choice between managing this marginal land to improve the habitat or to convert it to improved grassland should take into account both the benefits to biodiversity and ecosystem services and the risks that agricultural improvement will not be economically viable.;
- the SFS will be implemented at farm scale and almost all farms in Wales have both improved land and semi-natural habitats and features; future business decisions at farm scale will lead to inter-related, decisions about the management of both, especially on livestock and mixed farms;
- reversing the decline in biodiversity by improving the current condition and long-term resilience of existing semi-natural habitats is expected to require the creation of new semi-natural habitats, wetlands, and wooded land and features on improved and semi-improved farmland.

This review covers evidence for

- improved grassland management (infield and modified field margins)
- arable management (infield and modified field margins)
- organic farming
- management of agriculturally unproductive land and features
- significance of context, scale and synergy of biodiversity interventions on improved land

Although arable crops only occupy as small proportion of improved land in Wales and grassland management is by far the most significant land use, in some case the available evidence comes predominantly from studies of arable land, not improved grassland; this is noted where relevant below.

[16] www.erammp.wales/4
[17] When published, accessible at: https://erammp.wales/r-forest-evidence

8.1 Improved grassland management (infield and field margins)

- The increasing intensity of agricultural management of improved grassland in Wales led to the conversion of pastures and hay meadows to highly productive grasslands for silage in the second half of the 20th century and, more recently, wider use of short-term grass leys and more intensive use of enclosed farmland for sheep production.
- There is good evidence that adapting mowing or first grazing dates on improved or semi-improved grasslands can benefit wild plants and invertebrates, and also mammals when mowing is very infrequent. Certain bird species benefit from specific mowing patterns but studies have not shown clear effects for birds generally. For mobile species, some mowing effects may be critically influenced by the landscape context presence of refuge habitats nearby.
- There is good evidence that converting improved/semi-improved grassland to species-rich grassland benefits wild plants and invertebrates, and that the landscape context is important, especially proximity of existing semi-natural habitats, to achieve connectivity benefits and provide opportunities for species to colonise.
- There is good evidence that the creation of permanent grass buffer strips or field margins in arable land, and their subsequent management for biodiversity, leads to increased diversity of plants, pollinators and other invertebrates. Depending on their connectivity with other habitats and specific management these strips can provide valuable habitats for small mammals, brown hares, hedgehogs, common toads, grass snakes, common lizards and birds. There are fewer studies of grass margins in pastoral systems but good evidence of increased abundance and richness of invertebrates where grazing and nutrient inputs are excluded.
- For other interventions reviewed evidence of biodiversity impacts on improved land is either limited or equivocal, reflecting the focus of agricultural biodiversity research on semi-natural rather than improved grassland.

8.2 Arable management (infield and field margins)

- There is good evidence of benefits to invertebrates, including earthworms and other soil biota (as well as visiting pollinators in some cases) from the following interventions in arable crop management, used alone or synergistically: reduced herbicide and pesticide use; reduced tillage techniques; crop rotations (especially those with legumes); and cover crops. However, evidence of theoretical second order benefits of improved invertebrate food supplies for birds and other taxa, and of benefits for plants is limited, weak or inconsistent.
- There is good evidence of biodiversity benefits of two interventions in cereal stubble management. Leaving overwinter stubbles (not sprayed) until late winter has clear positive effects on birds and on brown hare populations, and most likely to be practicable where the rotation includes spring cropping. Combining unsprayed winter stubble with a spring fallow (so that the land is effectively in fallow for a year) provides additional benefits for plant diversity and vegetation structure and invertebrate diversity, while continuing to provide habitat for birds and brown hares.

- There is good evidence that fallow or unsown plots within an arable crop have positive effects on skylark and lapwing, and also on plants.
- Of the arable field margin management options reviewed, there was clear evidence of positive benefits for invertebrates of: conservation headlands (cropped but not sprayed); unharvested cereal crop headlands; and planting nectar/wildflower strips. There was limited or equivocal evidence of benefits for birds and plants of these options.

8.3 Organic farming

- Although there are many studies documenting the positive effects of organic farming systems on the abundance and diversity of birds, plants and invertebrates, much of this evidence is of limited relevance to policy decisions about biodiversity management interventions on farmland in Wales, for two reasons. Most of the replicated studies of ‘paired’ organic and conventional farms were of arable farms (or did not specify the farm type) so may not apply to intensive grassland systems in Wales. Secondly there is evidence that observed increases abundance and diversity of birds and bats on organic farms may be more closely associated with the diversity, structure and management of landscape features on the farms than with the organic management of crops or livestock.

8.4 Management of non- agricultural land and features

- The creation, restoration and management of farm ponds (many of which have become silted over) will benefit most amphibians and birds, and there is some evidence to indicate consequential benefits for terrestrial species richness locally, compared to unrestored ponds.
- Hedgerows and other wooded linear habitats on improved land provide habitats that are in strong contrast to adjacent intensively farmed grassland and arable land, and there is good evidence that hedgerows positively affect the richness and abundance of flora, invertebrates and birds, and increase landscape connectivity. Managing hedgerows and linear woody features for biodiversity will, for example, focus on width, structural variation, connectivity and the intensity of management of the adjacent improved land.
- Patches of agriculturally unproductive land and remnant habitats are sometimes embedded in improved farmland. Evidence of the benefits of actively managing these areas for biodiversity is currently unclear, and it is not possible to generalise on the effectiveness of habitat management interventions. Place-based assessment and tailoring of intervention packages, to focus on less fertile land (e.g. semi-improved grasslands) and proximity to existing semi-natural habitats, will be important.

8.5 Significance of context, synergy and scale of biodiversity interventions

- There is good evidence that the performance of agri-environment scheme measures on improved farmland varies with the landscape context, particularly the simplicity/complexity of the surrounding landscape in terms of the amount of semi-natural habitat nearby. This strongly suggests that the objectives, targeting and implementation of SFS biodiversity interventions on improved land should be adapted with respect to both surrounding landscape structure and target species groups.

- The interaction of multiple adjacent biodiversity interventions on neighbouring farms is also important, both for mobile species and to accumulate larger areas of biodiversity habitat, in the sense that "the whole is greater than the sum of the parts".
- There are also added benefits from synergistic effects of combining different interventions within the same field e.g. arable tillage and crop management, or field and crop margins with hedgerow management.

8.6 Synthesis of findings

A high-level summary of our conclusions is presented below following the approach used in the previous reviews.

Table 8.1 Key outcomes, benefits and critical concerns associated with supporting the diversification of swards in improved grasslands. The colour coding system is presented beneath the table.

Confidence	Intervention Name	Key Outcomes	Key Benefits	Critical concerns
Grassland infield management				
	Reduce fertilizer, pesticide or herbicide use; use organic rather than mineral fertilisers	Improved grassland habitat condition		Little potential for input reduction on improved grassland in Wales
	Adapt mowing or first grazing dates on improved or semi-improved grassland; use mowing techniques to reduce mortality; leave uncut patches in silage fields	Improved grassland habitat condition and structure	Wild plants and invertebrates, and some birds	Refuge habitat nearby has critical influences on some mowing effects. Positive effects of specific mowing patterns for some bird species (but not clear in all studies)
	Change grazing management, including mob grazing	Improved grassland habitat condition and structure		Little studied; limited evidence for mob grazing (but timing critical and could be strongly negative for some birds)
	Manipulate silage mowing height	Improved grassland habitat condition and structure		
	Convert improved/semi-improved grassland to species-rich permanent grassland	Improved grassland habitat condition and extent	Wild plants and invertebrates. May require intervention to decrease soil fertility	Proximity to existing semi-natural habitats important, for species and landscape scale benefits. In the case of semi-improved grasslands, critical to weigh up the relative economic and environmental benefits of conversion to species-rich or agricultural improvement

Confidence	Intervention Name	Key Outcomes	Key Benefits	Critical concerns
Arable land infield management				
	Reduce fertilizer, pesticide or herbicide use	Improved arable habitat condition	Pollinators, arthropods, soil biota	
	Use organic rather than mineral fertilizers; input other organic matter	Improved condition of arable soil habitats		Nutrient quantity more important than type
	Reduced tillage	Improved condition of arable soil habitats	Invertebrates, earthworms, soil biota, small mammals	Use in synergy with other arable infield and field margin interventions
	Introduce or modify crop/grassland rotations; companion cropping; undersow spring cereals; and cover/catch crops (including legumes)	Improve habitat heterogeneity of improved farmland landscapes	Increases in pollinators, arthropods, soil biota	Use in synergy with other arable infield and field margin interventions
	Leave overwinter stubbles, unsprayed, into late winter	Improve condition/extent of arable habitat	Birds, brown hare	Use in synergy with other arable infield and field margin interventions
	Leave overwinter stubbles unsprayed and follow with a spring fallow	Improve condition/extent of arable habitat	Birds, plants, invertebrates and mammals	
	Fallow/unsown plots	Improve condition/extent of arable habitat	Skylark, lapwing,	Use in synergy with other arable infield interventions
	Arable reversion to grassland	Habitat heterogeneity of improved farmland landscapes		Benefits depend on species-diversity of the grassland created
Organic farming				
	Organic farming	Improved farmland habitats/food resources for some taxa	Greater benefits where organic farms are located in homogenous landscapes	Much of the research is of arable farming outwith Wales. Observed benefits appear to be linked to the landscape structure and restrictions on use of pesticides and herbicides
Modified management of strips/plots around or within the field				
	Permanent grass buffer strips/margins along field edges or within fields	Habitat heterogeneity of improved farmland (grass and arable)	Birds, plants, invertebrates, mammals, reptiles	Exclusion of grazing and nutrients improved benefits in pastoral systems
	Conservation headlands (unsprayed crop); unharvested cereal headlands	Habitat heterogeneity of arable land	Invertebrates	Use in synergy with arable infield interventions
	Beetle banks	Habitat heterogeneity of arable land	Invertebrates	Use in synergy with arable infield interventions
	Plant nectar flower mixture/wildflower strips	Habitat heterogeneity of improved farmland (grass and arable)	Pollinators	Use in synergy with arable infield interventions

Confidence	Intervention Name	Key Outcomes	Key Benefits	Critical concerns
	Plant winter bird food/cover strips	Habitat heterogeneity of improved farmland (grass and arable)	Birds, possibly brown hare	Possibly more effective where few other seed sources (e.g. pastoral landscapes)
	Leave cultivated, uncropped margins	Habitat heterogeneity of arable land		
Management of unproductive land and features				
	Management of farm ponds	Improved farmland habitats/food resources for multiple taxa		
	Management of hedges and wooded linear features	Improve habitat condition and extent	See ERAMMP Report-4: *SFS Evidence Review Annex-4 Building Ecosystem Resilience*	
	Management of farmland trees and woodland	Improve habitat condition and extent, and landscape heterogeneity		
	Management of small areas of semi-natural habitats and features embedded within improved land			
	Creation of new woodland and agroforestry on improved farmland	See *ERAMMP Report-32: National Forest in Wales – Evidence Review*		
Other interventions				
	Provide supplementary food for birds or mammals			No supporting evidence inpractice
	Control predatory mammals and birds (foxes, crows, stoats and weasels)			Not yet developed or tested as a defined intervention
All improved land interventions				
	Skills interventions: - assessors - farmers and advisers	Appropriate application of measures to improve habitat condition and/or extend habitats	Targeting, and environmentally cost-effective delivery to supports habitat specific interventions	Assessors and advisers must have ability to communicate with farmers, ecology can be learnt. Farmer skills links to economic resilience.
	Introduce pilot result-based payment schemes for key interventions	Find out if can improve cost-effectiveness of delivery of habitat improvement/creation (e.g.Natural England's pilot scheme for nectar/wildflower strips)	Biodiversity Recognition (public and farmers') of the role of farmers in biodiversity management	New concept for farmers and delivery agencies, pilot schemes essential to test what works (and doesn't) and why, before using more widely. Not suitable for all habitats or tested for other objectives (soil, water). Usage to date has been mainly for 'higher level' habitat management.

Colour Key:

- Blue = well tested at multiple sites with outcomes consistent with accepted logic chain. No reasonable dis-benefits or practical limitations relating to successful implementation.
- Amber = agreement in the expert community there is an intervention logic chain which can be supported but either evidence is currently limited and/or there are some trade-offs or dis-benefits which the Welsh Government need to consider.
- Pink = either expert judgement does not support logic chain and/or whilst logic chain would suggest it should work there is evidence of one or more of the following:
    - its practical potential is limited due to a range of issues (e.g. beyond reasonable expectation of advisory support which can be supplied and/or highly variable outcome beyond current understanding or ability to target),
    - the outcome/benefit is so small in magnitude with few co-benefits that it may not be worth the administration costs,
    - there are significant trade-offs.

9 References

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