Literature reviews on the effects of farming practices associated with the CAP greening measures on climate and the environment

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This report is presented by the European Economic Interest Grouping:

-Alliance Environnement-
European Economic Interest Grouping

The authors of this report are:
Evelyn Underwood, IEEP
Geoff Radley, IEEP
Kaley Hart, IEEP
Graham Tucker, IEEP
Miriam Grace, IEEP


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EEIG ALLIANCE ENVIROMNEMENT is formed by the following companies:

**ORÉADE-BRÈCHE Sarl** and **IEEP**
64 chemin del prat - 31320 Auzeville France
11 Belgrave Road – London United-Kingdom
Tél. : + 33 5 61 73 62 62
Tél. : + 44 (0) 20 7799 2244
Fax : + 33 5 61 73 62 90
Fax : + 44 (0) 20 7799 2600
Mail : t.clement@oreade-breche.fr
Mail : CFroomberg@ieep.eu
Represented by:
Thierry CLEMENT
Represented by:
Claire FROOMBERG
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1 INTRODUCTION AND PROCESS FOLLOWED

This report contains a comprehensive identification and review of published peer-reviewed literature and other accessible information on the effects of farming practices that could be influenced by the CAP Pillar 1 greening measures on biodiversity, water, soils, climate mitigation and adaptation and ammonia emissions. These literature reviews were carried out for the ‘evaluation study of the payment for agricultural practices beneficial for the environment’ for DG Agriculture and Rural Development by Alliance Environnement and the Thünen Institute.

The reviews deliberately cover the range of factors that influence the ability of a particular type of farming practice to deliver these environmental and climate outcomes. This enables the identification of where the nature, location and timing of the management is critical to the delivery of these outcomes. It has been used to inform the analysis of the environmental and climate effectiveness of the greening measures.

The literature reviews have sought to systematically review the available peer-reviewed literature to ensure the most exhaustive coverage possible. A standardised template was produced for each environmental/climate issue. In summary, the process undertaken to identify the relevant information sources was as follows:

- A search through references cited by the reports and information sources that the study team knew to be of relevance from previous work in these fields, supplemented by a search of IEEP’s Reference Database and the references included in the literature review on the impacts of EFAs on biodiversity and ecosystem services carried out for JRC (Tzilivakis et al, 2015). IEEP’s Reference Database currently holds over 13,500 references, including at least:
  - 1,250 on agriculture and biodiversity;
  - 350 on agriculture and soils;
  - 350 on agriculture and water;
  - 350 on agriculture and climate.
- A search for references that cite the references found in the previous step, using Google Scholar.
- Systematic literature searches using Science Direct with various combinations of relevant key words (see Table 1) in the title, keywords or abstract
- A search for additional references by key authors and keywords, using Research Gate.

Although this process was carried out for all environmental issues the literature reviews differ in length. That for biodiversity is particularly long owing to the greater complexity of the topic – with the variations in effects of farm management practices on different species and habitats to be considered – than, say, soils or water quality. The length of the reviews also reflects the degree to which literature is available. For example, a detailed biodiversity review was feasible due to the fact that this is a topic which has been the subject of significant detailed research over the years.

---

1 Chapter 3 of Title III of Regulation (EU) 1307/2013
3 Each farming practice permitted under the greening measures and their equivalent practices, as defined in Annex IX of Regulation (EU) 1307/2013 is covered.
**Table 1: Key terms used for the systematic literature search by environmental issue:**

<table>
<thead>
<tr>
<th>Literature searches were carried out with combinations of the following key words in the title, keywords or abstract:</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Common terms for all environmental issues:</strong></td>
</tr>
<tr>
<td>• Crop diversity</td>
</tr>
<tr>
<td>• Crop diversification</td>
</tr>
<tr>
<td>• Crop rotation</td>
</tr>
<tr>
<td>• Buffer strip</td>
</tr>
<tr>
<td>• Fallow</td>
</tr>
<tr>
<td>• Cover crop</td>
</tr>
<tr>
<td>• Catch crop</td>
</tr>
<tr>
<td>• Green cover</td>
</tr>
<tr>
<td>• Woodland strip</td>
</tr>
<tr>
<td>• Landscape features</td>
</tr>
<tr>
<td>• Farmland features</td>
</tr>
<tr>
<td>• Field boundary</td>
</tr>
<tr>
<td>• Hedge</td>
</tr>
<tr>
<td>• Tree</td>
</tr>
<tr>
<td>• Copse</td>
</tr>
<tr>
<td>• Field margin</td>
</tr>
<tr>
<td>• Pond</td>
</tr>
<tr>
<td>• Ditch</td>
</tr>
<tr>
<td>• Terrace</td>
</tr>
<tr>
<td>• Afforested areas / afforestation</td>
</tr>
<tr>
<td>• Agro-forestry</td>
</tr>
<tr>
<td>• Short rotation coppice</td>
</tr>
<tr>
<td>• Nitrogen fixing crop</td>
</tr>
<tr>
<td>• Nitrogen fixation</td>
</tr>
<tr>
<td>• Arable</td>
</tr>
<tr>
<td>• Cereals</td>
</tr>
<tr>
<td>• Permanent grassland</td>
</tr>
<tr>
<td>• Semi-natural grassland</td>
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</tr>
<tr>
<td>Bee</td>
</tr>
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<td>Butterfly</td>
</tr>
<tr>
<td>Invertebrate</td>
</tr>
<tr>
<td>Mammal</td>
</tr>
<tr>
<td>Pollinator</td>
</tr>
<tr>
<td>Soil fauna</td>
</tr>
<tr>
<td>Flowering plants</td>
</tr>
<tr>
<td>Nectar and pollen</td>
</tr>
<tr>
<td>Legume</td>
</tr>
<tr>
<td>Alfalfa</td>
</tr>
<tr>
<td>Lucerne</td>
</tr>
<tr>
<td>Medicago sativa</td>
</tr>
<tr>
<td>Soya</td>
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<tr>
<td>Glycine max</td>
</tr>
<tr>
<td>Mustard</td>
</tr>
<tr>
<td>buckwheat</td>
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<td>Phacelia</td>
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<table>
<thead>
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<tr>
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</tr>
<tr>
<td>Soil conservation</td>
</tr>
<tr>
<td>Erosion</td>
</tr>
<tr>
<td>Soil organic carbon</td>
</tr>
<tr>
<td>Soil organic matter</td>
</tr>
<tr>
<td>Compaction</td>
</tr>
<tr>
<td>Salinisation</td>
</tr>
<tr>
<td>Contamination</td>
</tr>
<tr>
<td>Soil biodiversity</td>
</tr>
<tr>
<td>Water availability</td>
</tr>
<tr>
<td>Water quality</td>
</tr>
<tr>
<td>Surface water</td>
</tr>
<tr>
<td>Ground water</td>
</tr>
<tr>
<td>Eutrophication</td>
</tr>
<tr>
<td>Nitrate pollution</td>
</tr>
<tr>
<td>Nitrogen fixing crop</td>
</tr>
<tr>
<td>Soil organic carbon</td>
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<tr>
<td>Water erosion</td>
</tr>
<tr>
<td>Freshwater</td>
</tr>
<tr>
<td>Water bodies</td>
</tr>
<tr>
<td>Carbon dioxide</td>
</tr>
<tr>
<td>Methane</td>
</tr>
<tr>
<td>Slurry</td>
</tr>
<tr>
<td>Carbon dioxide</td>
</tr>
<tr>
<td>Ammonia</td>
</tr>
<tr>
<td>Soil organic carbon</td>
</tr>
<tr>
<td>Nitrous oxide</td>
</tr>
<tr>
<td>Carbon storage</td>
</tr>
<tr>
<td>Nitrous oxide</td>
</tr>
<tr>
<td>Carbon sequestration</td>
</tr>
<tr>
<td>Soil organic matter</td>
</tr>
<tr>
<td>Runoff</td>
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<tr>
<td>Phosphate pollution</td>
</tr>
<tr>
<td>Soil organic carbon</td>
</tr>
<tr>
<td>Erosion</td>
</tr>
<tr>
<td>Water availability</td>
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<td>Storms</td>
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<tr>
<td>Water availability</td>
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<td>drought</td>
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<tr>
<td>Climate adaptation</td>
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<td>Resilience</td>
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<td>Vulnerability</td>
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<tr>
<td>Storms</td>
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<tr>
<td>Climate adaptation</td>
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<tr>
<th><strong>Water</strong></th>
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<tbody>
<tr>
<td>Climate mitigation and adaptation</td>
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<tr>
<td>Ammonia emissions</td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th><strong>Climate mitigation and adaptation</strong></th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Ammonia emissions</strong></td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th><strong>Bird</strong></th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Soils</strong></td>
</tr>
<tr>
<td><strong>Water</strong></td>
</tr>
<tr>
<td><strong>Climate mitigation and adaptation</strong></td>
</tr>
<tr>
<td><strong>Ammonia emissions</strong></td>
</tr>
</tbody>
</table>
2 LITERATURE REVIEW ON THE EFFECTS ON BIODIVERSITY OF FARMING PRACTICES RELEVANT TO THE PILLAR 1 GREENING MEASURES

2.1 INTRODUCTION

Scope and coverage

Biological diversity is the variability among living organisms from all sources, including the diversity within species, between species and of ecosystems. This review considered the biodiversity between species within the farmland ecosystem at the field or farm scale (without touching on genetic diversity within species or ecosystem /landscape diversity). A simple count of numbers of species (species richness) is, however, a simplified and sometimes misleading view of biodiversity, as species have different levels of conservation importance (e.g. native species that are rare or otherwise threatened, restricted to certain habitats and concentrated or endemic to Europe are a high priority for conservation in the EU). This review therefore focuses on certain species groups that are a high priority for conservation, good indicators of the state of nature, and/or provide particularly important ecosystem services in the farmland environment.

This literature review focuses on biodiversity in the following categories of wild native species and other taxa, to the extent to which information is available:

- Wild native plants growing within fields (i.e. weeds) and/or in field margins, with a focus on flowering plants that provide nectar and pollen resources for invertebrates (bees, parasitoid wasps, flies, butterflies etc.) and plants that provide seeds or fruit for invertebrates, birds, mammals or other wildlife;
- Invertebrates, including:
  - soil macro-invertebrates of importance for maintain soil condition and as food resources for other species (e.g. earthworms and Collembola);
  - key invertebrate natural enemies of pests (e.g. weed seed predating and carnivorous Carabid beetles, spiders, predatory bugs, or parasitoid wasps);
  - butterflies and moths associated with farmland, i.e. excluding species whose larval food plants are not likely to be found in or around arable land;
  - pollinators (other than butterflies), including bumblebees, solitary bees, and hoverflies; the review does not consider honeybees;
- Common farmland birds, i.e. those included in the farmland bird index (EBCC et al, 2015);
- Species that are the focus of the Birds and Habitats Directives, i.e. those listed in Annex I of the Birds Directive and Annexes II and IV of the Habitats Directive that are particularly associated with farmland according to the EEA (EEA, 2015b).

Limitations to the review

To the extent possible the analysis consistently distinguished and quantified biodiversity impacts of the farming practice with respect to the biodiversity components listed above. When reviewing the relevant literature we paid particular attention to evidence of population impacts. However, this was significantly constrained because many research publications report on effects of agricultural crop choices or management on abundance (e.g. impact on observed foraging individuals or breeding pairs) but do not include data that can be used to infer a population level effect (Dicks, Showler and Sutherland, 2010). It is, for example, possible that some studies could be detecting a redistribution of individuals in the landscape in response to changes in resources, particularly studies of foraging butterflies and bumblebees (Power and Stout, 2011; Westphal, Steffan-Dewenter and Tscharntke, 2009), rather than an overall increase in survival and the reproductive rate that will increase populations.

---

4 Based on the definition in the Convention on Biological Diversity (CBD)
This review took into account the observation that a high proportion of species in farmland landscapes are generalist species, able to adapt to the high disturbance levels and highly modified habitats, as for example shown by farmland bird communities in the EU (Le Viol et al, 2012). Typically there are few habitat specialists present, and these are most likely to be rare and declining species on farmland. Furthermore, increasing the diversity of structures and habitats on farmland will generally increase the diversity of generalists, because of the wider range of niches available, but has more complex and species-specific impacts on specialist species, and can be detrimental to them. It is also important to consider the scale of changes in farmland landscape heterogeneity and resulting biodiversity impacts, as for the more wide-ranging species groups the effects of agricultural change operate at a landscape level and that examining species diversity at a local level does not reflect total species richness across an agricultural landscape (Hendrickx et al, 2007).

To take these issues into account, we paid particular attention, as far as available data allowed, to the impacts of farming practices on specialist farmland species, and threatened and declining species, rather than overall species diversity. We also looked for evidence of landscape and population scale impacts rather than local increases in the diversity of species or their abundance.

2.2 CROP DIVERSIFICATION GREENING MEASURE

Overview of potential farming effects and biodiversity impacts

The crop diversification measure may lead to biodiversity impacts through one of more of the following potentially interacting effects on farming systems and practices:

- The introduction of new crops (e.g. oil-seed rape, maize, beans, temporary grassland) or fallow land into arable dominated landscapes.
- Changes in the timing of the crop production cycle (as spring and autumn sown crops count as different crops), e.g. with spring sown cereals introduced into landscapes that are dominated by autumn-sown cereals, as is typical over most intensively managed farmland in the EU.
- Increases in crop rotation length or diversity (e.g. introduction of break crops, or more complex rotational systems).
- Changes in fertiliser and pesticide use as a result of the above changes. Break crops or more complex rotations may reduce use of fertilisers and some pesticides due to reduction in pest / pathogen populations or increase in soil organic matter. On the other hand, the introduction of some crops (e.g. oil-seed rape) can increase fungal disease risks and slug populations, leading to higher uses of fungicides and molluscicides in cereal crops.
- Other changes in farming practices, such as tillage methods or crop husbandry, as a result of the above changes.

These changed farming practices can in turn lead to a variety of complex and interacting impacts on biodiversity. This can result from differences in the ecological value of the individual crop types (e.g. as potential habitats for species), the impacts of their management, and the overall impact resulting from the increase in crop diversity across the farmland landscape. As these farming effects and their impacts interact, and are rarely studied in isolation, this review considers them together in relation to each focal taxa group. More detailed accounts of the biodiversity value of the most common nitrogen-fixing crops (legumes), cover and catch crops, and fallow land is provided in the EFA section.

It is important to note that the diversification measure is most likely to result in cropping changes in the most specialised arable farming areas, which tend to have the most intensively managed and largest fields. In contrast, small scale farms and mixed farming areas that are exempt from the requirement will have higher crop diversity, and tend to have smaller scale and less intensively manged fields, with higher non-farmed to farmed habitat ratios. The review of evidence therefore focuses on crop diversity studies that relate to the effects of crop diversity in typical intensive arable systems on large farms.

Pesticide use varies in different crops, but also depends greatly on the timing and type of cropping system and rotation (Box 2-1). In the UK, winter barley has higher average herbicide spraying frequency (2.6x) and fungicide spraying frequency (2.4x) than in spring barley (2.2x and 1.9x...
respectively) (see Table 2) Fungicide spraying frequency is notably high on potatoes, followed by wheat, rye and oilseed rape, compared to other arable crops. In Germany, pesticide spraying frequency averages are higher for winter oilseed rape than for winter wheat or barley, but the available data do not allow for comparisons between spring and winter cropping (see Table 3).

### Table 2: Herbicide, insecticide and fungicide spraying frequencies on arable crops in the UK (2014 for arable crops / 2013 for maize)

<table>
<thead>
<tr>
<th>crop type in UK</th>
<th>average herbicide spraying frequency on treated area</th>
<th>treated area</th>
<th>average fungicide spraying frequency on treated area</th>
<th>treated area</th>
<th>average insecticide spraying frequency (excluding seed treatments) on treated area</th>
<th>treated area</th>
</tr>
</thead>
<tbody>
<tr>
<td>sugar beet</td>
<td>5.1</td>
<td>100%</td>
<td>1.7</td>
<td>94.9%</td>
<td>1.3</td>
<td>13.3%</td>
</tr>
<tr>
<td>rye</td>
<td>2.1</td>
<td>100%</td>
<td>3.0</td>
<td>99.6%</td>
<td>1.3</td>
<td>78.5%</td>
</tr>
<tr>
<td>winter barley</td>
<td>2.6</td>
<td>99.2%</td>
<td>2.4</td>
<td>98.9%</td>
<td>1.2</td>
<td>67%</td>
</tr>
<tr>
<td>oilseed rape</td>
<td>3.7</td>
<td>98.4%</td>
<td>3.2</td>
<td>98.1%</td>
<td>2.4</td>
<td>83.1%</td>
</tr>
<tr>
<td>ware potatoes</td>
<td>3.3</td>
<td>98.0%</td>
<td>11.9</td>
<td>98.4%</td>
<td>1.8</td>
<td>50.6%</td>
</tr>
<tr>
<td>wheat</td>
<td>2.9</td>
<td>97.3%</td>
<td>3.7</td>
<td>98.4%</td>
<td>1.4</td>
<td>60.8%</td>
</tr>
<tr>
<td>maize (feed/silage)</td>
<td>2.0</td>
<td>97.1%</td>
<td>0.9</td>
<td>19.4%</td>
<td>0.9</td>
<td>6.4%</td>
</tr>
<tr>
<td>field peas (dry)</td>
<td>3.2</td>
<td>96.8%</td>
<td>1.5</td>
<td>82.2%</td>
<td>2.6</td>
<td>88.8%</td>
</tr>
<tr>
<td>spring barley</td>
<td>2.2</td>
<td>96.3%</td>
<td>1.9</td>
<td>90.5%</td>
<td>1.1</td>
<td>19.5%</td>
</tr>
<tr>
<td>faba beans</td>
<td>2.8</td>
<td>95.9%</td>
<td>1.8</td>
<td>87.9%</td>
<td>2.6</td>
<td>87.7%</td>
</tr>
<tr>
<td>oats</td>
<td>2.0</td>
<td>92.4%</td>
<td>1.9</td>
<td>86.9%</td>
<td>1.1</td>
<td>49.6%</td>
</tr>
<tr>
<td>new leys – direct sown</td>
<td>1.2</td>
<td>45.6%</td>
<td>1</td>
<td>34.8%</td>
<td>1</td>
<td>6.9%</td>
</tr>
<tr>
<td>grassland 2-5 yrs old</td>
<td>1.2</td>
<td>26.9%</td>
<td>-</td>
<td>3%</td>
<td>1</td>
<td>2%</td>
</tr>
</tbody>
</table>

Source: (Garthwaite et al, 2014; Garthwaite et al, 2013)

### Table 3: Herbicide, insecticide and fungicide spraying frequency on arable crops in Germany

<table>
<thead>
<tr>
<th>crop type in Germany</th>
<th>average spraying (2014)</th>
<th>herbicide frequency</th>
<th>average spraying (2014)</th>
<th>fungicide frequency</th>
<th>average spraying (2014)</th>
<th>insecticide frequency</th>
</tr>
</thead>
<tbody>
<tr>
<td>winter wheat</td>
<td>1.8</td>
<td>2.6</td>
<td>0.7</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>winter barley</td>
<td>1.5</td>
<td>1.7</td>
<td>0.3</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>winter oilseed rape</td>
<td>2.3</td>
<td>2.9</td>
<td>2.7</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>maize</td>
<td>1.5</td>
<td>0.0</td>
<td>0.03</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>potatoes</td>
<td>2.4</td>
<td>7.6</td>
<td>0.8</td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

Source: (Rossberg, 2016) (NB note that these data cannot be directly compared to the UK data as the German spraying frequency may include fields in which no pesticides were applied)

### Box 2-1: Impacts of crop rotation and crop diversity on herbicide and fungicide use in Germany

A study of herbicide (glyphosate) use in North German arable farming over ten years found that use was closely associated with minimum tillage practices on larger farms (Andert, Bürger and Gerowitt, 2016). A ten year study of fungicide and herbicide use on crop rotations in North Germany found that use intensities were generally smaller in more diverse crop sequences. Diversified cereal sequences, involving roots and tubers, maize or spring cereals were less dependent on herbicides, whilst cultivation of maize in three subsequent years increased herbicide use (Andert et al, 2016).
The crop diversification measure is being implemented in the context of significant shifts in crop rotation patterns and crop diversity in parts of Europe. For example, a study which identified crop rotation patterns for a large and representative area in Northern Germany (federal state of Niedersachsen with 1.8 million ha of arable land) during a six-year period from 2005 to 2010 (Steinmann and Dobers, 2013), concluded that due to a rapid increase of maize area in the region, crop sequence patterns are undergoing a dramatic shift. As an overall average, 46.9% of the maize area was grown in monoculture (i.e. as succeeding crops) in the region. During the six year study period 36.4% of arable area was cultivated with winter crops in 5 to 6 out of the six years and 17.1% of arable area was devoted to spring sown crops in 5 to 6 years out of six.

**Wild native plants**

Weed density and species richness varies between different crops due to their different structures and cropping times. In general sparse and short crops tend to support more species as they allow some non-crops plants to survive. However, modern cultivars of many crops grown under their intended intensive conditions often grow so quickly they out-compete most non-crop plants. Spring sown crops tend to have higher weed diversity and density than autumn-sown crops (Dicks et al, 2013) (see Box 2-2), in particular they contain more plants that are important food resources for invertebrates that form the food for many farmland birds, bats and other wildlife (Hald, 1999). The switch to winter cereals in Denmark led to a dramatic decline in the weed flora over 20 years (Hald, 1999).

Overall, herbicide and fertiliser use are the major factor in determining the composition, diversity and abundance of weed flora, although other factors such as mechanisation also play a role (Andreasen and Streibig, 2011). Crop rotations using minimum tillage are more likely to use pre-and post-emergence broad spectrum herbicides, but other factors also shift the weed communities in crop rotations with minimum tillage and it is not possible to generalise its impacts on weed diversity (Hernández Plaza et al, 2011).

Break crops and crop rotations are generally used to try to control certain weed problems, as each crop type allows a different weed community to develop, thus potentially increasing weed diversity in the rotation. However, there appears to be little evidence that increasing crop diversity in conventional arable systems leads to increases in wild native plant diversity within fields (i.e. alpha diversity) or between fields (i.e. beta diversity) or across the landscape (i.e. gamma diversity). Studies that found higher plant species richness in farms with more complex rotations did not distinguish between the impact of crop diversity and of lower fertiliser use in the more complex rotations (Billeter et al, 2008; Hilbig 1997 reviewed in Dicks et al, 2013) (see Box 2-2 for details). Plant species richness in field boundaries increases with increasing area of semi-natural habitat in the landscape which is often correlated with higher crop diversity. A study of weed species richness in winter wheat in Germany found that whilst species richness was higher in the organic crop rotations there was no difference in species richness between conventional diverse and conventional simple crop rotations (Ulber et al, 2009). The authors indicate that the key factors increasing weed species richness were the absence of chemical herbicide use and nitrogen fertiliser in the organic system.

**Box 2-2 Evidence of the impact of crop diversity on plant species diversity / weed density**

*Spring cropping:* Comparative studies in Denmark and the UK found that planting crops in spring rather than autumn resulted in higher weed diversity (Hald, 1999) or higher weed density (Jones et al. 1997 reviewed in (Dicks et al, 2013)). The study in Denmark compared the weed flora of unsprayed spring and winter cereals in 19 fields during 5 years, and found that plant and species densities, and accumulated species richness, were lower in winter than in spring cereals (Hald, 1999). Plant species and taxa that are important food resources for arthropod herbivores occurred at greater densities in spring than in winter cereals and, in addition, occurred with the highest relative abundance in spring cereals. For a few species the relative plant abundance was highest in winter cereals; these species were able to germinate both in the spring and autumn.

A large-scale comparison of 25 agricultural landscapes (of 16km2) in seven European countries (Billeter et al, 2008) found that crop diversity (average number of crops cultivated on a farm ranging from 1.2 to 7.7) was positively associated with non-woody vascular plant species richness (species number in the 4x4km2 study sites) in fields and field boundaries (semi-natural areas and linear elements). The study also found that vascular plant species richness increased with increasing area of semi-natural habitat in the landscape and decreased with increasing percentage of heavily fertilized agricultural land, and either semi-natural habitat or fertiliser use might be correlated with crop diversity. One study in Germany found higher plant species
Invertebrates

Sparse and short crops tend to support more invertebrate species as they allow some non-crop plants to survive, together with their associated herbivore invertebrates and their predators. Spring sown crops tend to contain more plants that are important food resources for invertebrates, higher weed diversity and density than autumn-sown crops (Dicks et al, 2013). Oil-seed-rape and other flowering crops that may be grown as break crops or in rotations, such as beans, provide food resources for nectar feeding species. However, the suitability of a crop for invertebrates is mostly dependent on the number, range and timing of pesticide applications, due to the direct mortality from insecticides, and the indirect impacts on the invertebrate food web that result from the loss of arable weeds following herbicide applications (Marshall et al, 2003).

There is relatively strong and widespread evidence that the diversity of invertebrates in farmland landscapes increases with habitat diversity and landscape heterogeneity (Benton, Vickery and Wilson, 2003; Weibull, Östman and Granqvist, 2003). There is considerable evidence that much of this effect is due to the diversity of non-cropped habitats, and especially semi-natural habitat, in the landscape (Benton, Vickery and Wilson, 2003; Billeter et al, 2008; Dainese et al, 2015; Firbank et al, 2008; Hendrickx et al, 2007; Le Féon et al, 2010; Oliver et al, 2010); however, most studies did not account separately for the influence of crop diversity.

The evidence of the impact of spatial and/or temporal crop diversity on invertebrate natural enemies and natural pest control is equivocal (see Box 2-3). A study on biological control of aphids in wheat in two different European locations, which untangled the influence of landscape complexity and crop diversity, found that pest control by natural predators increased with landscape complexity independently of the number of crops in the rotation (Rusch et al, 2013). The study found no simple relationship between crop rotation length and natural pest control. In longer and more diversified crop rotations, the overall level of natural pest control was more variable between sites, but parasitism rates were less variable (Rusch et al, 2013). However, landscape-wide crop diversification can reduce pest densities if the area of crops and other habitats that harbour the pest varies significantly from year to year (Bosem Baillod et al, 2017).

Box 2-3 Evidence of impact of diverse crop rotations on invertebrate natural enemies

The impact of crop rotations and crop diversity on invertebrate natural enemies is complex, as different species groups are affected in different ways. A study on biological control of aphids in wheat in two different European locations found that pest control by natural predators increases with landscape complexity, independently of crop rotation intensity, whilst simple crop rotations increased the within-field stability in natural control of aphids in wheat (Rusch et al, 2013). Another study of cereal aphid parasitism on winter wheat in northern Germany (Bosem Baillod et al, 2017) found that aphid densities were lower in landscapes where the cover of aphid host habitat had decreased from the year before, i.e. with increased crop diversity. The study concludes that landscape-wide crop diversification can reduce aphid densities. A study of cereal fields in western France (Bertrand, Burel and Baudry, 2016) found that total carabid abundance increased (and evenness between species decreased) with increase in temporal heterogeneity of the crop mosaic, most probably because of the increased dominance of a few generalist species, but there was no positive effect of crop diversity and mean field size on carabid species richness. Some species with high dispersal power such as Trechus quadristriatus were more abundant in landscapes with high spatial heterogeneity, whereas the abundance of less mobile species such as Poecilus cupreus was only positively influenced by temporal crop dynamics. A Hungarian study (Lövei 1984 cited in Dicks et al, 2013) found higher ground beetle species diversity in maize fields in crop rotation but a significantly higher abundance and activity of ground beetles in maize monoculture. A large-scale comparison of 25 agricultural landscapes (of 16 km²) in seven European countries found that crop diversity (average number of crops cultivated on a farm) was positively associated with the species richness of carabids and bugs, though the study did not distinguish between predatory and herbivorous carabid species and did not investigate whether changes in species richness influenced natural biological control of crop pests (Billeter et al, 2008). The study found that increased crop diversity was closely related to changes in fertiliser use and it is not clear which factor is key to the species richness effect.
Common farmland birds
Crop type, and in particular its structure, density and height, has an important influence on the suitability of field habitats for birds, both for nesting and feeding. Crops that are too tall and dense are difficult for birds to move within and to effectively forage. In contrast, crops that are too sparse may not provide adequate cover from predators. As a result spring sown crops tend to be more suitable for many ground-nesting species than winter-sown cereals, which are often too tall and dense by the time birds start breeding. Consequently, the widespread switch from spring-sown cereals to winter-sown cereals over much of North West Europe is considered to be one cause of the decline of farmland birds such as Stone Curlew, Corn Bunting, Skylark and Lapwing (Newton, 2004). As a result, some ground-nesting birds are known to preferentially nest in peas and spring bean crops, although nest failure rates tend to be high probably as a result of inadequate ground cover (see EFA section on nitrogen fixing crops). But it is important to note that crop nesting preferences vary according to regional farming practices and growing conditions. For example, in Finland and Sweden, traditional winter crops (mainly Rye) provide more suitable habitats for many species, including Skylark and Grey Partridge as they do not become too dense (Poláková et al, 2011). Sowing of spring sown crops coincides with the onset of some species breeding season, which results in the destruction of their nests. Consequently, fluctuations in Finnish Skylark populations appear to be linked to changes in the proportion of spring sown crops (Piha et al, 2007). In contrast, some species that normally nest in tall vegetation, such as reedbeds, have adapted to nesting in oilseed rape, for example Linnet and Reed Bunting (Moorcroft and Wilson, 1999; Newton, 2004).

Changes in crop type and in particular the timing of establishment also have important implications on their food resources and suitability as foraging habitat for birds. Spring sown crops tend to contain more plant, higher weed diversity and density and more invertebrates than autumn-sown crops (Dicks et al, 2013), which in turn can provide food for many farmland birds, as well as other wildlife (Hald, 1999). For example, cultivation of sugar beet in the UK, which is sown in late spring, is more likely to contain spring-germinating species such as Chenopodium album, a key food resource for some farmland birds (Holland et al, 2006).

The presence of crop stubble fields overwinter is of well documented importance for many seed-eating farmland birds, especially if they have high densities of crop weeds (Moorcroft et al, 2002). Consequently, switches from spring-sown crops to winter-sown crops, which has occurred in arable areas of Europe, results in the early ploughing of cereal stubble fields in autumn and winter, and deprives many farmland birds of key food resource seeds from the crop and from weeds (Geiger et al, 2010a). There is strong evidence from the UK that the loss of winter stubbles has contributed to farmland bird declines (Chamberlain et al, 2000; Newton, 2004; Whittingham et al, 2005). Stubbles are also important for wintering granivorous birds in Mediterranean Europe (Moreira et al, 2005).

There is evidence that crop diversity can affect the suitability of farmland for some ground nesting birds, especially if the crops are established at different times. This has been shown in studies of the Skylark in Sweden and the UK, where breeding success is so low on arable farmland that pairs must make two or three nesting attempts per year to maintain their population. However the height and density of vegetation within the agricultural fields constrains the amount of suitable habitat for nesting within each Skylark territory. Therefore, where the farmland has a high spatial and seasonal diversity of crop structure across a mosaic of small fields a higher density of Skylarks is able to make multiple nesting attempts than in landscapes with low crop diversity and large field sizes (Chamberlain et al, 1999; Schlaepfer 1988 cited by Eraud and Boutin, 2002; Wilson et al, 1997).

Crop diversity might be expected to positively correlate to overall bird species richness, as it is expected that more mobile taxa such as birds are influenced by composition and configuration of the habitat structure at larger spatial scales than more sedentary species with lower dispersal abilities (Tscharntke et al, 2005). Evidence in support of this is mixed. The diversity of farmland bird species has been found to be positively related to the heterogeneity of the main field types on a scale relevant to individual farms in Finland (Piha et al, 2007), and the Baltic States (Herzon et al, 2008). However, changes in bird populations were found to be unrelated to the level of farmland heterogeneity in Sweden (Wretenberg et al, 2007; Wretenberg, Pärt and Berg, 2010). Similarly, a recent study comparing species richness of birds (as well as spiders and butterflies) in relation to fine-
scale habitat land-use in intensive arable regions in Austria and Czech Republic found no association with habitat heterogeneity for any taxa (Gamero and Šálek, 2017). Instead, the study found that smaller patch sizes and larger areas of non-cropped elements (margins, hedges and grasslands) were associated with higher farmland biodiversity of the different taxonomic groups, farmland specialists and species of EU conservation concern.

**Box 2.4 Evidence of impacts of crop diversity on common farmland birds**

A study that accounted for non-crop elements in a mixed farming landscape in Sweden showed that crop structural diversity (i.e., the management and vegetation structure of crops) rather than crop diversity senso stricto positively affected richness of non-crop breeding bird species with stronger effects in arable, compared with forest-dominated landscapes (Josefsson et al, 2016). No such effects were observed among field-nesting farmland bird species. Another Swedish study found that species richness and territory abundance of ground-foraging species were significantly lower in autumn- than in spring-sown cereal plots both in the crop fields and the infield non-crop islands during the breeding season, whilst no such effect was observed among foliage gleaning birds (Eggers, Unell and Pärt, 2011).

A UK study (Henderson et al, 2009) showed a rapid and sustained population increase among a wide range of bird species in response to the replacement of winter wheat crops with oilseed rape, vining peas (*Pisum sativum*) and fallow with crop stubbles, natural regeneration of weeds and late spraying. Three species of UK conservation concern began breeding on the site that had not been present in the early years (i.e., Lapwing, Yellow Wagtail and Tree Sparrow).

A longterm study of bird species richness at the landscape scale in southern Portugal (Santana et al, 2017) found that the composition of the crop production component had the strongest effects on avian diversity, with a particularly marked effect on the richness of farmland and steppe bird species. Composition of the natural component was also influential, mainly affecting the richness of woodland/shrubland species. Although there were some effects of compositional and configurational heterogeneity, these were much weaker and inconsistent than those of landscape composition.

A UK study (Donald and Forrest, 1995) of Corn Bunting declines on arable farms found that crop diversity, hedgerow length, field size and the areas of all crops except wheat and barley did not differ significantly between years of highest and lowest Corn Bunting population. The authors conclude that Corn Bunting numbers are unlikely to have declined as a result of decreased crop diversity, or as a result of changes in the relative areas of tillage or grassland or because of the loss of hedgerows, and that more likely causes are reduced winter food supplies resulting particularly from the loss of spring tillage, increased pesticide usage and improved harvesting and storage techniques.

**Species that are the focus of the Birds and Habitats Directives**

Species that are the focus of the nature directives that occur on arable farmland are mostly specialists that are dependent on particular crops and semi-natural habitat elements and farming practices that are closely associated with low intensity arable farming systems. Such systems are therefore already diverse and are unlikely to be affected by the crop diversification measure.

### 2.3 MAINTENANCE OF PERMANENT GRASSLAND GREENING MEASURE

#### 2.3.1 NO PLOUGHING OF PERMANENT GRASSLAND

Permanent grasslands, and especially semi-natural grasslands, are species-rich ecosystems in Europe and have great conservation value, as indicated in Table 4. Semi-natural grasslands and grazed scrub and heath host species-rich vegetation communities, with up to 80 plant species per m²; some species are restricted to such habitats and dependant on specific agricultural practices (Veen et al, 2009), including many grassland butterflies (Van Swaay, Warren and Lois, 2006), amphibians (Temple and Cox, 2009) and reptiles (Cox and Temple, 2009) that are now threatened in Europe. The biodiversity value of semi-natural grasslands is evident from the inclusion of most of their types in Annex 1 of the Habitats Directive as habitats of Community interest. These habitats should therefore be the focus of conservation measures, including protection through Natura 2000 site designation and the adoption of appropriate conservation management measures, within Natura 2000 sites and the wider environment in order to maintain or restore their favourable conservation status. Semi-natural grasslands are also recognised as being a core component of High Nature Value (HNV) farming systems (Balock, 1999; Balock et al, 1993; Cooper et al, 2007; Keenleyside et al, 2014).

It is not possible to determine what proportion of EU permanent grassland is species-rich semi-natural grassland, but the area reported as being grassland defined under Annex 1 of the Habitats Directive as of European conservation interest covers at least 20% of permanent grassland in the EU (EEA, 2015a). The loss of extensively grazed semi-natural grassland is a major factor in the loss of biodiversity in Europe’s agricultural landscapes (EEA, 2010a), particularly the abandonment of extensive livestock systems on montane pastures and meadows, and areas with poor soils and harsh climates (Dover et al, 2011; Laiolo et al, 2004).
Table 4: Agricultural habitats in the EU, their importance for selected threatened habitats and species, and their overall biodiversity importance


<table>
<thead>
<tr>
<th>Habitat types</th>
<th>Natural habitats</th>
<th>Semi-natural habitats</th>
<th>Improved grassland</th>
<th>Cultivated</th>
<th>Crops</th>
<th>Permanent</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Pastures</td>
<td>Meadows</td>
<td>Organic</td>
<td>Conventional</td>
<td>Extensive</td>
<td>Organic</td>
</tr>
<tr>
<td>HD Annex 1 habitats*1</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>63</td>
<td></td>
</tr>
<tr>
<td>BD Annex 1 species*2</td>
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<td></td>
<td></td>
<td></td>
<td>54</td>
<td>32</td>
</tr>
<tr>
<td>European HD Annex II Butterflies*3</td>
<td>9</td>
<td>25</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td>European threatened amphibians*4</td>
<td>3</td>
<td>5</td>
<td>0</td>
<td>1</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td>European threatened reptiles*5</td>
<td>1</td>
<td>4</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td>Overall biodiversity importance</td>
<td>Very high, many species are restricted to such habitats</td>
<td>Very high, these habitats tend to be species-rich and declining; some species are restricted to such habitats and dependant on specific agricultural practices</td>
<td>Moderate, species diversity is much reduced compared to natural and semi-natural habitats, but some bird species of conservation importance use such habitats, sometimes in important numbers</td>
<td>High, such habitats are now rare and support some threatened species (esp birds)</td>
<td>Low, especially in intensive non-organic farmland dominated landscapes, but biodiversity levels can be enhanced by appropriate measures</td>
<td>Moderate - High, such habitats are declining and support some threatened species</td>
</tr>
</tbody>
</table>
2.3.2 The Impacts of Agricultural Improvement on Grasslands

There is clear evidence that the process of agricultural improving grasslands leads to a consistent and profound decline in their biodiversity conservation value (Goriup, 1988; Küster and Keenleyside, 2009; Poschlod, Baumann and Karlik, 2009; Stoate et al, 2009; Veen et al, 2009; Wallis De Vries and van Swaay, 2009). This is often a gradual transition from semi-natural extensively used grasslands to improved grasslands, and as a result some ecologists refer to semi-improved grasslands. These may be improved, such as through the use of manure, but are lightly grazed and therefore, although they are no longer considered to be semi-natural, they may be species-rich rich in terms of plants, invertebrates and birds. The drainage, use of fertilisers and herbicides on grasslands leads to their further agricultural improvement, which may culminate in their ploughing and reseeding etc. Most lowland grasslands in the EU have been subject to such improvements and are at least an order of magnitude lower in their biodiversity value than semi-natural and natural habitats (and do not include any Annex I habitats).

The biodiversity value of permanent grassland is particularly affected by the amount of time that has gone by since it was last ploughed and reseeded. Ploughing has a major effect on the soil invertebrates, as a result of the mortality from the physical action itself and exposure to predators, reductions in soil organic matter and insulating vegetation (Edwards, Hendrix and Arancon, 2012). Consequently, earthworm and adult beetle densities have been found to be positively associated with the age of the grassland (i.e., since it was last ploughed, and probably as a result are favoured feeding habitats by invertebrate feeding birds (e.g. Starling, Rook and Golden Plover) in winter (Tucker, 1992). However, the factors affecting use of grasslands by birds is complex as the sward height affects the accessibility of food resources and this may be more critical than lower food abundance with increased management intensity (Atkinson et al, 2005).

Reseeding of grassland with a few selected grass cultivars adapted to high nutrient conditions has a profound impact on the vegetation and associated fauna. It greatly reduces plant species diversity (and associated animal communities), increases the density and growth rates of the grassland, and impacts soil biodiversity. Many such improved grasslands have ten or fewer plant species in the sward, and are dominated by improved cultivars of Ryegrass (Lolium perenne), Cocksfoot (Dactylis glomerata), and White Clover (Trifolium repens). The most intensive temporary grasslands, such as grown for silage are often grass monocultures with no higher plants of conservation value present at all. Furthermore, the recovery of species-rich grassland takes decades, as for example shown by a study of botanical composition of grasslands that had regenerated following cultivation in Germany (Waldhardt and Otte, 2003). This found that the flora and vegetation are clearly related to the age of the grasslands with stands over 48 years old having the highest species richness.

The reduction in plant diversity from ploughing and reseeding has major knock-on impacts on their associated fauna. For example, the ploughing and reseeding of old pastures with rye-grass swards eliminates all known larval food plants of British butterflies (Boatman et al, 2007).

Ploughing and reseeding of grasslands is also normally accompanied by the use of high levels of fertiliser, which also often results in the replacement of hay cutting and/or grazing with silage production, where the forage is cut early, stored and then taken to animals that are held in high density stockyards. The grass is intentionally cut before seeding to maximise its nutritional value and this results in an absence of food resources for seeding birds etc. (Buckingham et al, 2010). The loss of grazing livestock from grassland converted to silage production is detrimental for biodiversity because animal dung (particularly from cattle) supports invertebrates, some of which are important prey for several species of bat (Wickramasinghe et al, 2003) and many birds (Vickery et al, 2001).

The early cutting of silage is also a major problem for ground-nesting birds, which results in very low rates of breeding success for many species (Buckingham, Giovannini and Peach, 2015). Furthermore, other mechanical operations on intensively managed grasslands frequently occur, including the spreading of fertiliser, topping of vegetation as weed control and rolling. These cutting and other operations are so frequent that there is insufficient time for breeding in between, and therefore egg
and chick mortality rates are so high such grasslands become sink habitats, i.e. they attract birds but breeding success is lower than mortality rates (Buckingham, Peach and Fox, 2006).

Despite their considerably reduced biodiversity value compared to semi-natural grasslands, some intensively managed grasslands do support some widespread and adaptable species, particularly birds (Tucker, 1992; Tucker and Evans, 1997). For example, some improved grasslands are favoured feeding sites for some internationally important wintering populations of geese and swans (which require protection under the Birds Directive).

2.4  **ECOLOGICAL FOCUS AREAS**

2.4.1  **LAND LYING FALLOW**

EFA fallow must be maintained without production for around half a year, from January or March to a cut-off date at the end of June, July or August (varying between countries or regions according to implementation rules). As all types of fallow are allowed (within different restrictions), including temporary grassland, with no particular incentives to farmers to adopt one type over another, the literature review considered four ways in which fallow can be managed:

- Bare (tilled) and natural regeneration of vegetation
- Crop stubbles with natural regeneration of vegetation
- Sown with plant mixes designed for environmental and biodiversity benefits
- Temporary grassland (sown before the EFA fallow period commences)

It is not expected that fallow land is treated with insecticides or fungicides, as this would not be an economically rational way to deal with potential pest or disease problems that might carry over to crops, but it may be treated with herbicide or topped or mulched to control problematic weeds where this is allowed by Member State EFA rules. As fallow land on EFAs cannot be tilled or otherwise disturbed during the fallow period, and pesticide treatments are generally limited to spot treatment of problematic weeds and a final herbicide burn-off of vegetation before ploughing, it generally offers broadly greater biodiversity benefits than the nitrogen-fixing crops or catch and cover crop options. It should be noted, however, that cereal stubbles that are ploughed and left fallow as bare soil are highly prone to soil erosion (Boellstorff and Benito, 2005), which may cause biodiversity losses off-site, for example in aquatic habitats.

There is a large body of evidence of the biodiversity benefits of fallow land. However, most of it refers to land that is left fallow for a year or several years, whilst on EFAs Member State rules have defined that the fallow lasts only 5-8 months from the start of the year or from March. A meta-analysis of the impacts of fallow (as set-aside) concluded that land withdrawn from conventional production unequivocally enhances biodiversity of birds, insects, spiders and harvestmen, and plants in Europe (Van Buskirk and Willi, 2004). The study found that benefits are greatest on large parcels of land, and on older fallow for all taxa except bird species richness, which declined significantly with increasing age of fallow. A review in the UK found that farmland birds benefited from (unsprayed) one-year fallow (rotational set-aside) more than multi-year fallow, whilst small mammals benefited mostly from multi-year fallow (Silcock and Lovegrove, 2007). In Germany, a field assessment of various potential EFA options ranked multi-year fallow with natural regeneration highest for biodiversity benefit to weeds, flower-visiting invertebrates, and farmland birds, followed by multi-year sown flowering seed mixes (fallow or margins) and annual sown flowering margins or annual fallow with natural regeneration. All the options were managed with no application of pesticides and fertilisers, no soil cultivation before 30 November, and no cutting between 15 April and 1 August (Dziewiaty et al, 2013). A review of the impacts of long-term fallow (as set-aside) on biodiversity (Tscharntke, Batáry and Dormann, 2011) concluded that the impact is highest in simple landscapes,

5 The duration of this rule is not defined in the regulation, and as a result of a request by the UK, the Commission allowed the planting of crops after mid-summer
where improvements have the highest relative effect and are influenced by sowing patterns and age of succession, whereas in complex landscapes fallows cannot add much to an already high biodiversity. However, fallows in more complex or extensively managed landscapes can provide key resources for some species of conservation concern.

**Wild native plants**

Fallow with natural regeneration of vegetation generally provides a relatively rich plant diversity after one to two growing seasons, including plant species that are Lepidopteran larval food plants, that provide nectar and pollen resources for flower visiting insects, and plants that provide food for farmland birds (Boatman et al, 2011; Van Buskirk and Willi, 2004). Fallow sown with species-rich seed mixes also tends to reach maximum plant species richness in the second year (Tscharntke, Batáry and Dormann, 2011). Fallow sown with temporary grassland will generally only provide plant diversity that supports other wildlife if the grass species mix is made up of less competitive species and/or in areas with arable crop rotations that have received low levels of nitrogen fertiliser (Kovács-Hostyánszki et al, 2011a; Kuussaari, Hyvönen and Härmä, 2011).

**Box 2-5 Evidence of wild plant abundance and species diversity on fallow**

<table>
<thead>
<tr>
<th>Bare (tilled) and natural regeneration of vegetation</th>
</tr>
</thead>
<tbody>
<tr>
<td>Bare tilled fallow is a conservation measure for rare arable weeds, but it is only effective on the soil types and cropping systems that still contain substantial seed banks of the threatened species (Höft, 2012).</td>
</tr>
</tbody>
</table>

**Crop stubbles with natural regeneration of vegetation**

A survey of long-term set-aside fallow in the UK (Boatman et al, 2011) found that although few sites had developed plant communities of significant conservation interest after several years, over 40% of plant species present after the first few years were larval food plants for butterflies and over 60% were food plants for farmland birds. An earlier review in the UK found that plant species richness was greater on natural regeneration than on sown grass covers, though the species richness, particularly broad-leaved weeds, was significantly poorer in intensive arable landscapes than in areas with mixed farming (Firbank et al, 2003). A study in Finland (Kuussaari, Hyvönen and Härmä, 2011) found that plant species richness was highest in the second year of fallows with unsown stubbles and in fallow sown with less competitive grasses, compared with fallow sown with a competitive grass-clover mix. In a German study, weed species richness was much higher on one-year old fallow plots with natural regeneration of vegetation than in field pea crops, but fallow plots sown with *Phacelia tanacetifolia* had fewer species than field pea crops (Gathmann, Greiler and Tscharntke, 1994).

**Sown with plant mixes designed for environmental and biodiversity benefits**

Studies found that fallows sown with species-rich seed mixes tend to reach maximum plant species richness in the second year when both annuals and perennials are present (Tscharntke, Batáry and Dormann, 2011), whereas 3 to 4 year old fallows can become dominated by perennial grasses and therefore have lower plant species richness (Toivonen, Herzon and Helenius, 2013).

**Sown with temporary grassland**

Fallow fields (one to three years old) sown with a legume (usually alfalfa) and two grass species in arable farming areas in Hungary had higher plant species richness and insect-pollinated plant species richness compared to neighbouring winter cereal fields after one year, in some cases even exceeding the plant species richness in semi-natural grasslands in the same landscapes (Kovács-Hostyánszki et al, 2011b). The fallow was not treated with herbicide or fertiliser and mown once in the second half of June. The fields had previously received only moderate levels of fertiliser, with an average of 70kg N/ha/yr applied to winter wheat crops in the area.

**Invertebrates**

Fallow can be expected to increase soil invertebrate macro-fauna abundance primarily because of the lack of disturbance from tillage and pesticides, compared to arable soils (Krogh et al, 2007; Nieminen et al, 2011). The effects increase over time in long-term fallow (Kautz et al, 2010), as the diversity and the biomass of soil organisms, including bacteria, fungi or earthworms, is generally significantly lower in arable soils compared to areas with permanent vegetation (de Vries et al, 2013). For some groups, the differences may only be apparent after more than one season of fallow (Tóth et al, 2016).

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6 However, this may be influenced by the fact that the pea fields had been left to grow for at least 6 months before the start of plant sampling, whilst the *Phacelia* was sown at the start of the plant surveying period.
Fallow fields (one to three years old) sown with a legume (usually alfalfa) and two grass species in arable farming areas in Hungary had significantly higher butterfly abundance and species richness compared to neighbouring winter cereal fields (Kovács-Hostyánszki et al, 2011b). Species composition was dominated by the large and highly mobile pierid species whose larvae feed on Brassicaceae species which were abundant on the fallow. The fallow was not treated with herbicide or fertiliser and had previously received only moderate levels of fertiliser, with an average of 70kg N/ha/yr applied to winter wheat crops in the area. A study in Finland (Kuussaari, Hyvönen and Härmä, 2011) found that butterfly and day-active moth species richness was significantly higher in fallow sown with less competitive grasses (Agrostis capillaris-Festuca ovina) and in fallow with stubbles than the competitive grasses fallow (Festuca pratensis-Phleum pratense-Trifolium pratense). Lepidopteran species richness was higher after two years than in the one-year fallow. However, no lepidopteran species significantly preferred the fallow plots over the field margin habitats.

There is evidence that fallow with natural regeneration of vegetation can rapidly provide flower resources for wild bees and hoverflies, and also provides attractive nesting habitat for solitary bees (Gauthmann, Greiler and Tscharntke, 1994). However, destruction of EFA fallow after half a year will destroy bee nests, and it does not therefore offer nesting habitat for solitary bees. Fallows sown with diverse seed mixes containing flowering plant species also quickly benefit foraging bumblebee numbers (Alaen et al, 2011), with some evidence that field-size (1 ha) patches may have a population-level effect (Carvell et al, 2015). Well-chosen seed mixtures can improve the availability of nectar and pollen sources in short-term fallow as most arable fields have small seedbanks lacking in perennial broad-leaved weeds which provide the most important resources (Alaen et al, 2011). See field margins section below for a discussion of the evidence that small-scale sown seed mixes (field margin strips and patches) benefit pollinators.
Box 2-8 Evidence of impacts of fallow on pollinators – wild bees and hoverflies

Crop stubbles with natural regeneration of vegetation

Fallow with naturally developed vegetation (two years old) had twice as many cavity-nesting solitary bee species as sown crops, whereas the predatory species (wasps and parasitoids) showed a rather uniform distribution between the fields (Gathmann, Greiler and Tscharntke, 1994). Management by cutting greatly increased plant species richness in early-successional set-aside fields and thus doubled species richness of solitary bees.

Sown with plant mixes designed for environmental and biodiversity benefits

Bumblebees were monitored on long-term fallow (6 years) sown with three different seed mixtures and mown or not mown compared to surrounding field margins in Finland (Alanen et al, 2011). Bumblebees showed a very strong positive response to the diverse seed mixture with abundant floral resources, and their abundance peaked in the first year. Short-tongued bumblebees were very abundant in the year of establishment on Phacelia tanacetifolia, whilst long-tongued species increased during succession. Bumblebees were systematically more abundant on the fallow than on field margins. A UK study (Heard et al, 2007) showed that legume-grass flowering patches of 0.25, 0.5 and 1 ha sown with a mixture of 20% legumes (Trifolium pratense, Trifolium hybridum and Lotus corniculatus) and 80% fine-leaved grasses (Festuca rubra, Poa pratensis and Cynosurus cristatus) attracted significantly higher densities of bumblebees than patches of non-crop vegetation typical of the site (average 26 bumblebees/200 m2 on forage patches compared to 2 bumblebees/200 m2 on control patches). A large-scale study of the same legume-grass flowering patches found that the larger patches (1 ha) may be increasing the population level of bumblebee species in the landscape over more than one season. The biomass of bumblebee males and queens was higher in landscapes surrounding larger (1 ha) than smaller (0.25 ha) sown patches, although the effect of the presence of flower patches on biomass of bumblebee males and queens overall was not significant (Carvell et al, 2015).

Sown with temporary grassland

A study in Finland (Kuussaari, Hyvönen and Härmä, 2011) found that sowing one to two year fallows with a grass seed mixture with less competitive grasses (Agrostis capillaris-Festuca ovina) increased bumblebee species richness in the fallow, compared to fallow sown with a grass-clover seed mix of competitive species (Festuca pratensis-Phleum pratense-Trifolium pratense). Bumblebee abundance was highest in the stubbles and the less competitive grasses mix where the species richness of flowering plants was highest, though it should be noted that the surveying was carried out before the clover started to flower.

Common farmland birds

There is strong evidence from the UK that cereal stubbles on winter fallows and set-aside (as it was) provide valuable foraging habitat for farmland birds (Dicks et al, 2013; Gillings et al, 2010). Stubble fields left after harvesting of cereal crops in the autumn provide an important food source for granivorous farmland birds, which feed on both spilt grain and weed seeds in the soil (Moorcroft et al, 2002), and there is some evidence of subsequent positive effects on breeding abundance (Gillings et al, 2005). Another UK study found that farms with over 10% of the arable land under fallow had an average 16–53% higher abundance of key farmland birds than farms with less than 3% under fallow (Henderson et al, 2012). There is also strong evidence that fields or field margins sown with wild bird cover seed mixes provide key food resources for farmland birds, most importantly during the winter (Dicks et al, 2013; Vickery, Feber and Fuller, 2009).

Sowing fallow with temporary grassland seed mixtures based on competitive grasses produces a dense sward which is of low value for most foraging birds (Henderson, Vickery and Fuller, 2000). But it can provide breeding habitat for some grassland birds if the sward structure is suitable (Chamberlain, Vickery and Gough, 2000; Henderson, Vickery and Fuller, 2000; Herzon et al, 2011). Small unsown fallow patches within winter-sown cereals and summer fallow can provide good breeding habitats for farmland and grassland birds such as Skylark and Lapwing (Natural England, 2009). Although the surface area of these patches is too small to make a significant contribution to the overall EFA, the studies show the value of small fallow patches as targeted interventions to benefit particular species. However, the destruction of EFA fallow in mid-summer for planting cuts short the breeding period of some ground-nesting species, potentially leading to losses of eggs and young. The EFA may therefore become an ecological trap for such species, especially if they are attempting to rear a second or third brood.
Box 2-9 Evidence of impacts of fallows on farmland birds

A study (Henderson et al, 2012) across 28 farms in England found that sites with less than 3% (and, to a lesser extent, <5%) of fallow land were highly under-populated by birds (compared to national annual mean densities), whereas a two-fold increase in the area of fallow land (to more than 10%) was associated with an average 16–53% increase in the relative abundance of key bird species. The fallow land was either floristically enhanced temporary grassland, wild bird seed mix sown as winter cover, insect-rich cereal-vetch seed mix sown as summer cover, or stubble with natural regeneration, either as strips or as blocks, sown in spring or autumn and maintained for a year (Holland et al, 2013).

Crop stubbles with natural regeneration of vegetation

A systematic review (Dicks et al, 2013) found evidence in seven studies and reviews from the UK that leaving overwinter stubbles leads to higher densities of farmland birds in winter, increased Grey Partridge (Perdix perdix) productivity, and increased Cirl Bunting (Emberiza cirlus) population size (in combination with several other conservation measures) and territory density (Aebischer, Green and Evans, 1999). Seed-eating birds, in particular Linnet (Carduelis cannabina), Grey Partridge (Perdix perdix), Chaffinch (Fringilla coelebs), Yellowhammer (Emberiza citrinella), Reed Bunting (Emberiza schoeniclus) and Corn Bunting (Emberiza calandra) all benefit from overwinter stubbles that have a substantial weed flora and bare ground (Moorcroft et al, 2002). An analysis of data from 30 UK studies on set-aside stubble fields (Gillings et al, 2010) demonstrated that winter farmland bird densities tended to be higher on stubbles than on either cereal or oilseed rape crops. Small unsown fallow patches (4mx4m) in winter-sown cereals provided spring and summer nesting habitats for Lapwing (Vanellus vanellus) (Natural England, 2009) and Skylark (Morris et al, 2004).

Sown with plant mixes designed for environmental and biodiversity benefits

A systematic review (Dicks et al, 2013) identified that 21 comparative studies out of 31 from the UK and France found positive effects on birds of sowing wild bird seed or cover mixture on fields or field margins. Ten studies and four reviews from the UK found that fields sown with wild bird cover mix had higher abundance, density, species diversity and species richness of birds than other farmland habitats, recommending a seed-mix combination of kale, quinoa and seeding cereals. Nine replicated studies from France and the UK reported mixed or negative effects of wild bird cover on birds compared to other farmland habitats.

Sown with temporary grassland

A study in Finland (Herzon et al, 2011) found that long-term grass-sown set-aside fallow supported 25 to 40% more bird species and held 60 to 105% more pairs of birds typical of open farmland in comparison with cereal fields within a similar landscape setting. However, a UK study found that bird abundances were significantly higher on unsown set-aside than on temporary grassland fallow and winter cereals, and all bird functional groups were most abundant on short-term (one year) unsown fallow except crows (which preferred grassland) (Henderson, Vickery and Fuller, 2000).

Species that are the focus of the Birds and Habitats Directives

Fallow land can provide feeding, breeding and refuge habitat for species of conservation concern if it is in the right location at the right time of year (see Box 2-10). For example, there is evidence that Lesser Kestrel (Catry et al, 2012), Little Bustard (Santangeli and Dolman, 2011), Great Bustard (Rocha, Morales and Moreira, 2012) and the European Hamster (Fischer and Wagner, 2016) benefit from fallows in cereal rotations, either stubbles with natural regeneration of vegetation or planted with targeted wildflower seed mixes. However, the value of fallow as foraging habitat for birds may be reduced if it becomes too tall and dense. Hence, for example, Montagu’s Harriers have been found to preferentially hunt in alfalfa rather than fallow (Schlaich et al, 2015).

Box 2-10 Evidence of impacts of fallows on species that are the focus of the Birds and Habitats Directives

Stone Curlew (Burhinus oedicnemus)

Small fallow plots (4mx4m) in winter-sown cereals provide spring and summer nesting habitat for Stone Curlew in England (Natural England, 2009).

Lesser Kestrel (Falco naumanni)

A model of the foraging decisions of Lesser Kestrels in a cereal steppe landscape in Spain showed that the location of cereal and fallow patches within a 2-km radius of a kestrel colony influences the total food supply delivered to the nestlings, explaining the differences in breeding success between years and colonies (Catry et al, 2012). A decrease in the proportion of fallow fields within the cereal rotations is predicted to negatively influence breeding success, but the field harvesting sequence can play an important role in alleviating the effects.

Montagu’s Harrier (Circus pygargus)

Montagu’s Harriers caught the largest numbers of voles prey on cut strips in alfalfa, where prey were more available than in the fallow with tall vegetation, but fallow had highest vole abundance (Schlaich et al, 2015).

Great Bustard (Otis tarda)

Great Bustard in southern Portugal showed a clear preference for nesting in cereal fields, followed by young fallows, old fallows, ploughed fields and then other habitats (Rocha, Morales and Moreira, 2012).

Little Bustard (Tetrao tetrax)

A study of displaying male Little Bustards in a pastoral landscape on Sardinia (Santangeli and Dolman, 2011) found that they were influenced by the cover of legumes and green herbs on fallow and grassland, by short vegetation structure on grasslands and by distance from roads. Little Bustard males in southern Portugal were associated with vegetation with high floristic
Literature reviews on the effects of farming practices associated with the CAP greening measures on climate and the environment

| Richness and high abundance of legume species (Faria, Rabaça and Morales, 2012). |
| Hamster (Cricetus cricetus) |
| Long-term (5 year) sown wildflower fields in Germany (Fischer and Wagner, 2016) (with seed mixtures containing annual and perennial wild and cultivated plants, intermixed with e.g. sunflowers, fennel, mallow and chicory), and with no applications of pesticides, synthetic fertilizers, tillage and mowing, contained higher densities of hamster burrows than any other habitat type within the arable farmland. |

2.4.2 Terraces

EFA terraces must include those identified under cross-compliance (GAEC7) and can also include other terraces as permitted under Article 45. If Member States choose to permit terraces additional to those protected under GAEC7 then they must define criteria for these, including minimum height, to reflect local or regional characteristics. For example, Hungary and Romania chose to allow additional terraces of minimum 1m in height (Hart, 2015).

Terraces are raised, flat-topped banks of earth, created to control erosion and provide flat surfaces for farming on sloping land, and traditional terraces are usually constructed with a retaining dry-stone wall, although modern mechanically constructed terraces often have no retaining wall or use concrete (Farmer et al, 2008). Most terraces are constructed for the cultivation of permanent crops, but some arable land is found on terraces, including silvo-arable agroforestry systems. However, most field terraces are no longer in use for arable cropping as they are normally very narrow, which makes it impossible to work on them with large farm machinery.

Stone-walled terraces typical of Mediterranean regions will have similar biodiversity values to those of stone walls, whereas the biodiversity benefits of grass terraces in more northerly regions will be similar to those of buffer strips, fallow land or unutilised agricultural areas, depending on management (Farmer et al, 2008). The evidence of the biodiversity benefits of terraces comes from terraces with permanent crops, but it was assumed in this review that terraces with arable crops could have similar benefits if located in the same regions.

Wild native plants

Stone-walled or earth banked terraces can provide disturbance free habitats with specific microclimates (either dryness and high sun exposure, or shadiness and moisture), which can provide micro-habitats for plant species that do not otherwise occur on farmland. For example, a case study found that Maltese stone-walled terrace landscapes support a number of endemic species, including Maltese spurge (Euphorbia melitensis), Maltese fleabane (Chiliadenus bocconei), Maltese spider orchid (Ophrys melitensis) and Maltese sea lavender (Limonium melitensis) (Rolé, 2007 cited in Farmer et al, 2008). Studies of the biodiversity value of terraces in the (Italian) Alpine Region showed terraced vineyards to be biodiversity rich and this value is dependent on low input farming methods (Lasen and Fagarazzi 2008 cited in Farmer et al, 2008).

Invertebrates

Intact terraces are expected to protect soil from erosion and runoff, and therefore will enhance soil macro-invertebrate populations compared to unterraced sloping arable soils. No direct evidence was found of this. For further evidence, see stone walls.

Common farmland birds

See stone walls.

Species that are the focus of the Birds and Habitats Directives

See stone walls.
2.4.3 LANDSCAPE FEATURES

Member States are able to choose whether to define landscape features according to their own GAEC standard or Article 45, or a combination of both. According to the EU regulation, an EFA landscape feature must be ‘at the disposal’ of the farmer and can be outside the eligible arable area if it is adjacent to it on at least one long edge. Member States have interpreted adjacent to the cultivated arable area in different ways, either allowing or not allowing a gap or another landscape feature between the arable area and the feature.

2.4.3.1 Hedgerows and wooded strips

Article 45 defines a maximum hedge and wooded strip width of 10m (gaps can be a maximum of 2m). Some examples of diverging rules defined by Member States are: in Belgium-Flanders gaps can be up to 4m. UK-England and UK-Wales define a minimum length of 20m, UK-Northern Ireland minimum 5m and gaps up to 5m, but the hedge can only be maximum 2m wide from the centre at the base. In the Netherlands and UK-Wales, hedges must be maximum 10m wide.

There is a very large and wide-ranging evidence base that shows that hedgerows and other woody field boundary habitats benefit wildlife by providing habitats, feeding sites, refuges, and movement corridors for invertebrates, birds, mammals, reptiles and amphibians, and they support some species that would not otherwise exist in arable landscapes (Farmer et al., 2008). The research indicates that hedges vary greatly in their character, with old hedges containing rich mixes of woody and herbaceous plant species and diverse habitats in trees, shrubs, internal gaps, bank, basal vegetation and bordering margins offering the greatest biodiversity value. Other woody field boundary habitats include lines of Mediterranean scrub and maquis, and copses and small woody patches within fields or in field corners. It is important to recognise that hedge and woody habitat communities are very different to those of farmland fields, and are often closer to those of scrubland and woodland. Therefore, hedge and wooded strip conservation tends to benefit different species than other elements used for implementation of EFA complementing in-field conservation rather than being an alternative.

Wild native plants

Hedges contain greater herbaceous plant species richness than crop areas, as well as a range of woody species (Marshall and Moonen, 2002). They also usually contain a higher herbaceous plant species richness than grassy field margins of comparable age (Pywell et al., 2005). Hedge and forest corridor specialist plant species tend to be shade tolerant generalists, which have the potential to be forest dwellers, but also use various alternative and perturbed shaded habitats in the landscape. The hedge plant flora therefore shares most species in common with tree lines and woody patches on farmland (see below), but contain only a narrow range of the plant community types found in larger areas of woodland (McCollin et al., 2000).

Invertebrates

Soil macro-invertebrates are likely to be much more abundant in hedges and other woody margins, which are undisturbed (i.e. un-tilled), than in crops subject to regular tillage (Nieminen et al., 2011). Species composition and richness is also likely to be significantly greater under hedges and woody borders than in crops because of the diverse vegetation structure with tree and shrub roots both above and below ground.

7 Article 46(2)c of Regulation (EU) No 1307/2013 of the European Parliament and of the Council
There is evidence that hedges are refuges for invertebrate natural enemies of crop pests, including predatory carabid beetles and spiders (Pywell et al, 2005), hoverflies with aphidophagous larvae, other predatory flies (Pfister et al, 2017), and parasitoid wasps (Macfadyen et al, 2011), which spill over into crops once pest populations develop (Dainese et al, 2016; Haenke et al, 2014; Inclán et al, 2016). A systematic review (Wright et al, 2013) of four studies (one from Europe) that measured the impact of planting hedges on arable farms on natural biological control found some evidence of benefits for natural enemies, although impacts were affected by the type of vegetation planted and between natural enemy groups; however none of the reviewed studies looked at pest regulation.

**Box 2-11 Evidence of impacts of hedges on invertebrate natural enemies of crop pests**

One study in northern Italy found that hedges increased cereal aphid parasitism in winter wheat crops (Dainese et al, 2016) compared to fields with no hedges. Adult syrphid abundance (of aphidophagous species) in hedges and forest edges adjacent to oilseed rape or winter wheat fields in Germany (Haenke et al, 2014) were more abundant in forest-connected hedgerows than in forest edges (with isolated hedges being intermediate), and more abundant in crop fields adjacent to hedgerows than adjacent to forest edges, indicating spillover from semi-natural habitats to the adjacent crop fields. A study in Italy, Germany and Switzerland found that adult predatory fly abundance in borders of arable fields was highest in hedges compared to herbaceous field margins (Pfister et al, 2017). A study that compared nearby organic and conventional mixed arable-pasture-livestock farms in SW England found that hedgerows produced more parasitoid species, significantly more so on organic farms, compared to grass fields and grassy field margins (Macfadyen et al, 2011). The greater parasitoid species richness on organic farms was linked to temporal stability in parasitism rates, but the analysis showed no consistent differences between the organic and conventional farm pairs in terms of pest control service (Macfadyen et al, 2009).

Hedges and woody strips are key foraging and dispersal habitats for butterflies and moths in arable farmland, as well as providing larval food plants in the hedge base vegetation so long as it is wide enough (Feber et al, 2007). In the UK it is estimated that around 40% of resident or regular migrant butterfly species potentially breed in hedgerows (Dover and Sparks, 2000). Farmland that is rich in hedges and tree patches hosts some Lepidopteran species that are typical of forest as well as the common farmland species, greatly increasing species richness in impoverished arable Lepidoptera communities (Belfrage, Björklund and Salomonsson, 2015; Dainese et al, 2015). Hedges with trees and extended width margins have increased Lepidopteran species richness compared to hedgered without these features (Merckx et al, 2012). However, hedges generally have a lower moth species richness compared to patches of woodland (see below) as they lack woodland specialists (Fuentes-Montemayor et al, 2012). Hedge management has a strong impact on butterfly and moth abundance, as well as the abundance of flower and berry resources for pollinators and birds, especially a low frequency of cutting (once every three years) and cutting in winter rather than autumn (Staley et al, 2016; Staley et al, 2012). However, many arable farmers choose to cut immediately after harvest when no crops are present to restrict access, immediately before sowing winter crops, whilst cutting in winter is often limited by poor access across wet soils.

**Box 2-12 Evidence of impacts of hedges and woody strips on Lepidoptera**

Presence of hedgerow trees (usually oak Quercus robur) locally increased macro-moth species richness, but not abundance, in predominantly arable UK lowland farmland (Merckx et al, 2012). The effect of hedgerow trees was particularly strong for shrub- and/or tree-feeding species. The study also found that extended-width tussocky grass field margins increased macro-moth species richness but not abundance, independently of the presence of hedgerow trees. However, the positive impacts of hedgerow trees and extended-width margins on macro-moth species richness were not relatively greater with an increasing amount of arable land in the landscape. A study of moth species abundance in woodland patches of different sizes within farmland in Scotland, UK found that woody strips with a high proportion of edge to the surrounding matrix were low in species richness because of the lack of woodland specialists (Fuentes-Montemayor et al, 2012). **Hedge management** by cutting only once every three years in winter increased Lepidoptera abundance and the diversity of components of the Lepidoptera community linked with specific lifecycle traits, and resulted in 2.1 times more flowers and a 3.4 times greater berry mass over 5 years compared to annual trimming in autumn in a UK study (Staley et al, 2016; Staley et al, 2012).

Hedges can provide stable hibernation and nesting sites for wild bees provided the base of the hedge is wide enough to include sunny patches for ground nesters, and old trees provide cavities for tree nesters, although sunny herbaceous field margins and fallow are preferred by ground nesting bumblebees (Lye et al, 2009). A study in Mediterranean arable landscapes in northern Italy (Dainese et al, 2016) found that increasing hedgerow cover in the landscape from 1 to 6% enhanced pollinator flower-visitation rate (bumblebees, solitary bees, hoverflies, other flies, butterflies, and other species) along the field margins. Hedges can also provide foraging resources from flowering plants in the
hedge base and from the woody trees and shrubs and creeping plants such as brambles, roses and ivy (Jacobs et al, 2009), which are particularly important in early spring or autumn for solitary bees (Wood, Holland and Goulson, 2016b) and bumblebee queens. The value of hedges for pollinators is strongly influenced by hedge and hedge base width, the timing and frequency of hedge trimming and the management of hedge base vegetation. A study in the UK found that bumblebee movement along hedges is strongly influenced by their connectivity, and more connected hedges had higher pollination rates of flowers preferred by bumblebees (Cranmer, McCollin and Ollerton, 2012).

**Common farmland birds**

There is a considerable amount of evidence that in structurally simplified arable landscapes, hedges provide one of the most important on-farm habitats for birds (Hinsley and Bellamy, 2000), and increasing hedge length significantly increases bird species richness (Batáry, Matthiesen and Tscharntke, 2010) (see Box 2-13).

However, it is important to bear in mind that such increases in species mainly relate to the addition of generalist species that are more typical of woodland and scrub habitats, than farmland specialists. In fact some open land specialist bird species (such as some larks) are deterred by the presence of hedges and woody field boundaries (as they can hold and hide predators) and are less abundant when they are present. On the other hand, some farmland species (such as some game birds, buntings and finches), do rely on hedges as nesting sites in farming landscapes that lack semi-natural vegetation. Therefore, in most situations farmland bird communities require a mix of habitats, including shrubby and woody vegetation in hedges and trees, and suitable open field habitats.

**Box 2-13 Evidence of impacts of hedges on common farmland birds on arable land**

Hedge length around both winter wheat fields and meadows enhanced bird species richness in simple landscapes (<17% semi-natural areas within a 500m radius around the centre of bird survey plots) across Germany (Batáry, Matthiesen and Tscharntke, 2010). In more complex landscapes the local effect of hedge length levelled off because bird richness was high even without local hedges. More bird species occurred in organic than in conventional fields regardless of land-use type (wheat fields and meadows), but hedge length had a much stronger effect on bird richness. A study in agricultural landscapes of SW Poland found that bird species richness and bird densities were significantly related to field margin aggregations and arranged along a decreasing gradient of landscape heterogeneity: shrubby mosaic, open mosaic, open plain plots (Wuczyński, 2016). Most species and breeding pairs were preferentially associated with shrubby margins.

**Species that are the focus of the Birds and Habitats Directives**

Hedges and woody strips are key breeding and feeding habitats for a few Annex I bird species such as Red-backed Shrike (*Lanius collurio*) (Brambilla, Rubolini and Guidali, 2007) and Lesser Grey Shrike (*Lanius minor*).

Numerous bat species also forage along hedgerows and tree lines (European Commission and Eurobats, 2014), including the Habitats Directive Annex II species *Rhinolophus hipposideros* and *Barbastella barbastellus*. Bats are highly sensitive to losses of such habitat features in farmland (Frey-Ehrenbold et al, 2013; Pocock and Jennings, 2008). Hedgerow length was positively associated with *Pipistrellus* spp foraging abundance in England (Boughey et al, 2011), and a review of conservation priority bat species in the UK found that they are dependent on both the tree and shrub component of the hedge and the field margin or ditch (Wolton et al, 2013).

A study in Spain found hedges to be a key habitat for the Egyptian mongoose *Herpestes ichneumon* and the common genet *Genetta genetta* (Pereira and Rodríguez, 2010). Other Habitats Directive Annex II species associated with hedges are the European Ratsnake *Zamenis situla*, and Mouse-tailed Dormouse *Myomimus roachi*.

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2.4.3.2 Isolated trees / Trees in a line

Isolated trees must have a crown diameter of minimum 4 m and trees in a line must have a crown diameter of minimum 4 m and the space between the crowns must not exceed 5 m, according to Article 45. However, Member States can include trees with a smaller crown diameter if they are recognised as valuable landscape features in that country. For example, Italy\textsuperscript{12} and France have defined tree species whose crown can be smaller than 4m.

Mature trees scattered throughout agricultural landscapes are critical habitat for some species but are declining rapidly in European landscapes (Gibbons et al, 2008).

\textbf{Wild native plants}

No evidence found – see above re woody strips.

\textbf{Invertebrates}

No evidence found – see above for evidence related to hedges and woody strips.

\textbf{Common farmland birds}

Individual boundary trees and in-field trees can be beneficial to more generalist bird species, but not many specialists (Eglington and Noble, 2010). They provide nesting habitat and song-posts for birds (Siriwardena, Cooke and Sutherland, 2012).

\textbf{Species that are the focus of the Birds and Habitats Directives}

Isolated mature trees can provide increased richness of tree-hole nesting birds and bats compared to tree less arable fields, as shown in studies in Australia (Manning, Gibbons and Lindenmayer, 2009). They may therefore be important for maintaining populations of some species that are the focus of the Birds and Habitats Directive on arable farmland, for example Hoopoe (\textit{Upupa epops}), and bats such as Noctule (\textit{Nyctalus noctula}) (Kalda, Kalda and Liira, 2015). Bats that forage in more open landscapes are likely to cluster around single trees but are more abundant around denser clusters of trees and areas of cluttered vegetation associated with high insect (mostly Diptera) abundance which could act as sources of prey (Fuentes-Montemayor et al, 2013). One study in Estonia found that bat flight activity was three times higher around double-tree-lines (alleys) than around single-tree-lines and solitary trees, and flight pattern was log-linearly related to tree density (Kalda, Kalda and Liira, 2015).

2.4.3.3 Trees in groups and field copses

The maximum area covered by EFA tree groups and field copses cannot exceed 0.3 ha according to Article 45. No other specifications are defined.

Small pockets of woodland are a fairly common feature on farmland in many EU countries. Distinctive elements of small woodland patches include a tree layer, a scrub layer and a herb layer. They can range from ancient native woodland, with extremely rich associated communities of flora and fauna, to new plantations of non-native species which support comparatively few native species. Farm woodlands have a disproportionately large value for species that utilise the woodland edge, due to the large edge length provided by small patches.

\textsuperscript{12} cypress and black poplar
**Wild native plants**

Tree groups and copses provide small patches of habitat for similar types of plant flora as found in hedges, but with more favourable conditions for woodland specialists (see above).

**Invertebrates**

Soil macro-invertebrates are likely to be more abundant in and around tree groups and field copses than in arable fields because of the lack of disturbance. Woodland edges are also likely to provide suitable refuge habitats for invertebrates on arable farmland (see evidence on hedges and forest edges).

Tree groups and copses are likely to provide refuges for invertebrate crop pest predators and parasitoids with a refuge from disturbance in the crop, a more favourable micro-climate (e.g. in dry sunny weather) and food resources from nectar, pollen, plant secretions or aphid honey dew. One study found woodland edges next to arable farmland increased abundance of two families of adult predatory flies in field margins (Pfister et al, 2017), and two studies found that tree patches increased abundance of parasitoids in olive plantations (see Box 2-14).

**Box 2-14 Evidence of the impact of woody patches on invertebrate natural enemies of crop pests**

A study in Italy, Germany and Switzerland found that adult predatory fly abundance of Daggerflies (Empididae) and Hoverflies (Syrphidae) in borders of arable fields was highest in woodland edges compared to herbaceous field margins; whilst Long-legged Flies (Dolichopodidae) were least abundant in woodland (Pfister et al, 2017). A study in an intensively managed olive plantation in southern Spain found that increased density of patches of herbaceous and woody vegetation within and around olive groves was linked to decreased abundance of key pests (Olive Psyllid and Olive Moth) in the crop (Paredes et al, 2013), and that the presence of herbaceous and woody vegetation patches was linked to increased abundance of natural enemies (parasitoids, spiders and heteropteran bugs) (Paredes, Cayuela and Campos, 2013). A study in traditional extensively managed olive plantations in Italy detected significant effects of woodland habitat from 0.5 to 2 km around the olive groves on parasitism by the complex of parasitoids of the Olive Fruit Fly (Bactrocera oleae) (Boccaccio and Petacchi, 2009). The degree of parasitism was highly variable across the six olive groves (ranging from 2.92 to 22.57%), but the study found the percent parasitoid emergence was significantly correlated with percent woodland area at 0.75 km, and negatively affected by the fragmentation of woodland at a spatial extent ranging from 1 to 2 km.

Tree groups and field copses can provide larval food plants and nectar for generalist butterfly species found on farmland, depending on the amount of forest edge and light habitat within the patch containing flower resources or flowering shrubs (Smart et al, 2000) (see Box 2-12). For example, willow species (Salix spp.) are key flowering resources in early spring. However, small and fragmented tree groups and copses are unlikely to contain a high species richness of Lepidoptera, which is driven by the absence of forest specialists that require larger areas of core woodland habitat (Fuentes-Montemayor et al, 2012).

**Box 2-15 Evidence of significance of wood patches for Lepidoptera**

A study of moth species abundance in woodland patches of different sizes within farmland in Scotland, UK found that moth abundance and richness were higher in large woodland patches located close to other woodlands (Fuentes-Montemayor et al, 2012). Small woodland patches can potentially maintain relatively high moth abundance and richness, if they have a relatively compact shape with less edge and are close to other woodlands. In general, woodlands composed of broadleaved trees only (no conifers) and a large number of tree species supported high moth abundance and species richness; there was a negative impact of grazing within the wood.

Forest edges adjacent to arable land can provide suitable undisturbed but sunny habitat for bee nesting, shady protected sites for hibernation, and wild plants for foraging (see forest edges for more detail). A Swiss study on grassland with differing isolation from forest edges and percentage of woody habitats in the surrounding landscape (500 m radius) found that the proportion of flowers setting seed was significantly reduced by isolation from forest due to less frequent visits by pollinating insects, and seed predation and insect scavenging by insect predators were significantly lower at isolated sites than at sites connected to woody habitat (Farwig et al, 2009). Although this study was on grassland rather than arable farmland, the effects of forest presence on arable farmland may be similar.
Common farmland birds
A study in Poland found the strongest positive predictor of species richness of all bird species, of species of European conservation concern and of farmland specialists was the length of woody edge habitat (composed of tree patches or tree lines) on farmland (Sanderson et al, 2009). A study of the abundance of 31 farmland bird species at the 1 km2 scale on UK farmland found a predominantly negative influence of large areas of woodland, but a predominantly positive influence of hedges and more heterogeneous landscapes (Siriwardena, Cooke and Sutherland, 2012). The study did not explicitly differentiate the effect of small areas of woodland (less than 0.5 ha), but it can be assumed that their effect is a component of the landscape heterogeneity effect rather than the woodland effect.

Species that are the focus of the Birds and Habitats Directives
Priority protected species associated with woodland edges on farmland include Grey Partridge (Perdix perdix) and bats (see Box 2-16). A review of European woodland bird population trends concluded that small pockets of woodland on farmland are likely to benefit declining species of European woodland birds, such as Lesser Spotted Woodpecker (Dryocopus martius) and Eurasian Wryneck (Jynx torquilla) (Gregory et al, 2007).

A study in Spain found riparian forest strips on farmland to be a key habitat for the Egyptian mongoose Herpestes ichneumon and the common genet Genetta (Pereira and Rodríguez, 2010).

Box 2-16 Evidence of impacts of woody patches on bats on farmland
A study that examined bat foraging activity and relative abundance (and insect prey availability) in woodland fragments in agricultural landscapes in the UK found that aerial hawker species (e.g. Pipistrellus species) had higher foraging activity in woodland with low tree densities and an open understory, whilst gleaning species (e.g. Myotis bats) showed the opposite trend (Fuentes-Montemayor et al, 2013).

2.4.3.4 Field margins
Field margins subject to cross-compliance must be retained through the year, but EFA field margins that are not protected by cross-compliance only have to be retained until the beginning of August. Obligatory GAEC field margins must have permanent herbaceous vegetation in order to serve as buffers to soil erosion, nutrient leaching and pesticide drift, so are generally covered mainly by grass species. In contrast, EFA margins that go beyond GAEC requirements can be cleared and sown with seed mixtures that benefit wildlife in some way. This review therefore considers that field margins can consist of:

- Permanent grass (with annual cut or more frequent cutting);
- Sown seed mixtures (pollinator seed mixes or game cover strips for birds or mammals);
- Bare unvegetated cultivated soil (allowed for example in Germany and Romania, but not in UK-Scotland).

Little information is available from Member States to characterise a ‘typical’ arable field margin. A review of field margins in 8 Member States concluded that grassy margins along water courses and hedges had increased as a result of cross-compliance rules except in Germany and Italy (Farmer et al, 2008). A survey of arable field boundary habitats in 39 regions in 10 European countries in summer 2014 recorded that field margins covered 0.1% - 1.1% of the arable land area with median widths of 1.1 - 5.5 m in different regions (IFAB, 2015). Evidence from two intensive agricultural regions in Germany in 2008-9 indicated that in one region around 80% of field margin length was narrow grassy strips (1-3m wide), and that these form a large proportion of the semi-natural habitat
in arable areas, whilst in the other region around half of the margin length was wider than 3m and included hedges and/or trees (Hahn, Lenhardt and Brühl, 2014).

Field margins and buffer strips can be a valuable habitat on farmland because they provide permanent relatively undisturbed habitat, and generally contain a wider range and diversity of species than the cropped area (Marshall and Moonen, 2002). Their value as habitats is greatly increased if they border onto another structurally rich habitat such as a hedge or ditch with riparian vegetation. Field margins up to a few metres from the crop edge are affected by herbicide and fertiliser drift (Boutin et al, 2014; de Jong, de Snoo and van de Zande, 2008; Dise, 2011; Prosser et al, 2016; Storkey et al, 2012), but they also buffer other bordering habitats such as hedges, ditches and wetlands from the negative impacts of crop management practices, especially fertiliser and pesticide applications.

**Wild native plants**

Field edges with their reduced management intensity and increased immigration have higher plant species richness than arable crops independently of the crop type and management (Batáry et al, 2012). The plant flora of field margins is dominated by species unique to the boundary habitats beyond the margin (Cordeau, Reboud and Chauvel, 2010; Marshall and Moonen, 2002). At the same time, field margins still provide a habitat for many arable weed species of conservation value which have disappeared or seriously decreased in the field core (Fried et al, 2009). However, many narrow field margins are heavily affected by fertiliser and pesticide drift from the crop, which results in the dominance of competitive-ruderal plant species and reduces overall plant diversity (Marshall and Moonen, 2002). Bare (cultivated) field margins provide habitat for rare arable plants in those fields where they are still present in the seed bank.

**Box 2-17 Evidence of the importance of field margins for wild plant diversity**

*Unvegetated (cultivated) field margins*

Uncropped annually cultivated field margins provide habitat for rare arable plants (Pywell et al, 2012).

*Sown field margins versus grassy margins*

The vegetation of buffer strips in England (Crichtley et al, 2013) established by species-rich seed mixtures or natural regeneration had greater value for wildlife than those established with a simple grass seed mixture, with greater bumblebee food plant richness, diversity of food plants for farmland birds and butterfly larvae, and perennial forbs important for invertebrates. Another UK study (Marshall, West and Kleijn, 2006) found that the herbaceous flora of the pre-existing field margin adjacent to sown 6 m margin strips was significantly more species-rich than controls, probably reflecting the way the additional sown strip buffered the impact of herbicide drift from the crop.

**Invertebrates**

Soil macro-invertebrates are much more abundant in undisturbed (i.e. untilled) field margins and other off-field habitats compared to crops (Nieminen et al, 2011) including legume crops (Smith, Potts and Eggleton, 2008). In contrast, field margins that are sown with seed mixes are likely to have similar soil macro-fauna abundances and species composition as neighbouring crops, as the soil disturbance from tillage and herbicide treatments will have significant negative impacts on most soil macrofauna groups (Smith et al, 2008).

**Box 2-18 Evidence of the importance of field margins for soil macro-invertebrates**

*Grass field margins*

A UK study (Smith, Potts and Eggleton, 2008) found that earthworms (Lumbricidae), woodlice (Isopoda), and rove beetles (Staphylinidae), as well as the three main soil macro-invertebrate feeding groups (litter consumers, soil ingesters and predators) had higher abundance and species density in sown grass strips compared with the faba bean crop. The species composition of soil macrofaunal communities in grass strips was different compared with the crop, the hedge next to the field margin, and with other habitats on the farm. Another UK study (Smith et al, 2008) of soil invertebrate macrofauna under sown four year old field margins found that diversity in the field margins was higher than in the crop, with earthworms, woodlice and beetles having significantly more species and/or higher abundances in the margins. However, scarification of the margins resulted in lower abundances and fewer woodlice species (Isopods) and also reduced soil- and litter-feeder abundances and predator species densities, although populations appeared to recover by the autumn, probably as a result of dispersal from neighbouring plots and boundary features. The species composition of the scarified margins was similar to that of the crop. Arable field margins in Finland harboured over twice the density and almost double the number of earthworm species in comparison to the adjacent cultivated fields, and eight of the nine species had wider regional distribution in the margins than in the fields (Nieminen et al, 2011).
Permanent vegetated field margins are refuges for invertebrate natural enemies of crop pests, from which they spill over into crops once pest populations develop (Bianchi, Booij and Tscharntke, 2006; Inclán et al, 2016). As tillage frequency and timing is one of the most important factors influencing predatory beetle abundance in crops, field margins are key for breeding, overwintering and/or dispersal (Holland and Luff, 2000). There is also evidence that beetle banks - raised earth banks across fields planted with grasses – increase predatory beetle activity in crop fields by providing overwintering habitat and a refuge from disturbance from which beetles can spill over into the field when pests become available (Collins et al, 2002; Wright et al, 2013) (see also section on buffer strips). Sown wildflower field margins also attract ground beetles and spiders (Aviron et al, 2009), but wild bird or game cover margins tend to host a lower diversity of predatory invertebrates than permanent (uncropped) margins (Vickery, Feber and Fuller, 2009).

### Box 2-19 Evidence of the importance of field margins for invertebrate natural enemies of crop pests

<table>
<thead>
<tr>
<th>Sown wildflower or wild bird/game cover margins</th>
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<tr>
<td>A comparison in Switzerland (Aviron et al, 2009) found wildflower strips sown with 20–40 species contained significantly more (8–60% more) ground beetle (Carabidae) and spider (Araneae) species than crop fields in the same region, but did not compare with other margin types. A review concluded that temporary margins with natural regeneration can provide similar benefits to species rich permanent margins, but only once sufficient diverse vegetation with structural complexity and flowering resources have developed, whilst sown wild bird or game cover margins rarely support as high predatory invertebrate diversity as permanent (uncropped) margins (Vickery, Feber and Fuller, 2009).</td>
</tr>
<tr>
<td><strong>Grassy margins</strong></td>
</tr>
<tr>
<td>A review and a UK study (Collins et al, 2002; Wright et al, 2013) showed that beetle banks - raised earth banks across fields planted with grasses – increase predatory beetle activity in crop fields by providing overwintering habitat and a refuge from disturbance from which beetles can spill over into the field when pests become available. However, effects on ground beetle species diversity may take a number of years to appear (Irmiler, Sommer and Neumann, 2015). Another review concluded that simple grassy margins provide overwintering habitat for predatory Carabid and Staphylinid Beetles, whilst more species rich sown permanent margins also provide habitat for spiders and nectar and pollen resources for predatory and parasitoid wasps, beetles, flies and bugs (Vickery, Feber and Fuller, 2009). Levels of cereal aphid control by epigeal and aerial natural enemies in winter wheat were positively related to the proportion of linear grass margins in a UK study (Holland et al, 2012a). A study in Italy (Inclán et al, 2016) showed that the spillover of tachinids (Diptera: Tachinidae) into maize crops was higher from grass margins than from hedgerows, suggesting that the spillover of this group may be related to the low contrast between the vegetation structure of the margin and the crop, whilst aphidophagous hoverfly predators (Diptera: Syrphidae) were abundant in crop centres next to all field margins. A study found that increasing the quantity and connectivity of grassy field margins increased wasp predation and parasitism of caterpillars (Holzschuh, Steffan-Dewenter and Tscharntke, 2009).</td>
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Permanent field margins can be key habitats for Lepidoptera on arable farmland if they are wide and have a high diversity of wild grasses and flowering plants that serve as larval plants (Fuentes-Montemayor, Goulson and Park, 2011; Merckx et al, 2012).

### Box 2-20 Evidence of the importance of field margins for butterflies and moths

<table>
<thead>
<tr>
<th>Sown wildflower margins</th>
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<tr>
<td>A comparison in Switzerland (Aviron et al, 2009) found wildflower strips sown with 20–40 species contained significantly more butterfly (Lepidoptera) species than crop fields in the same region. A UK study found a positive relationship between butterfly species diversity and the area covered by uncropped sown field margin strips at the farm scale (Holland et al, 2013). Another UK comparison found that the abundance and species richness of micromoths was significantly higher within field margins under wildlife friendly management in comparison to conventionally managed margins (Fuentes-Montemayor, Goulson and Park, 2011). In contrast, hedgerows under wildlife-friendly management enhanced neither micromoth nor macromoth populations.</td>
</tr>
<tr>
<td><strong>Grassy field margins</strong></td>
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</tbody>
</table>
| A UK study (Merckx et al, 2012) found that extended-width tussocky grass field margins increased macro-moth species richness but not abundance, compared to GAEC standard 1m wide mown grass field margins. A study in an arable landscape in France (Delattre, Vernon and Burel, 2013) found that Meadow Brown (Maniola jurtina L.) butterfly movement was facilitated by

arable landscapes with dense networks of grassy field margins compared to those without margins. A study in France found that butterfly species diversity along field edges in a mixed farming landscape was positively correlated to the surface area in linear herbaceous elements (road verges and hedge banks), and they provided higher butterfly diversity than grasslands which are more intensively managed and receive higher inputs than field margins (Ouin and Burel, 2002). However, a larger study in 18 landscapes in three regions of France found that butterfly communities were of lower conservation value, richness and abundance than in grassland patches and decreased with isolation from grasslands, indicating that although grassy linear elements probably play an important role in butterfly dispersal in agricultural landscapes, they may not be sufficient to preserve specialist and sedentary species (Villemey et al, 2015).

There is ample evidence that field margins sown with pollen- and nectar-rich plants enhance pollinator richness, although their effectiveness varies with the magnitude of increase in flowering plant cover resulting from the practices, farmland type, and landscape context (Scheper et al, 2013). A systematic review found that 50 comparative studies out of 65 in northern Europe14 (Dicks et al, 2013) showed some benefits of wildflower strips to one or more wildlife groups. Common species are the main beneficiaries of the establishment of wildflower strips, but there is some evidence that margins sown with pollen- and nectar-rich plants also attract rare species (Pywell et al, 2012). There is less evidence that sown flowering strips benefit solitary bees, many of which require resources in the spring (Wood, Holland and Goulson, 2016b).

Grassy margins and naturally regenerated margins support lower insect abundances and diversity than in sown pollen- and nectar-rich flower mixtures and other wildflower strips (Carvell et al, 2007; Marshall, West and Kleijn, 2006). However, grassy margins can provide undisturbed bee nesting and hibernation sites if they are not too frequently mown but contain some bare or sparsely vegetated patches and some flowering plant resources. Hoverflies with aphidophagous larvae may favour grass margins because of their importance for larvae (Holland et al, 2015).

It is important to note that all these studies measured the attraction of sown flower mixes to foraging invertebrates, and the abundance of foraging invertebrates on the strips may not necessarily represent an increase in the populations as they may have moved from wild flowers in other farmland habitats. Even if the overall population is temporarily increased, this may not result in an enduring effect on the species population if it does not translate into an increased number of emerging bumblebee queens or solitary bee nests. However, a recent UK study (Wood, Holland and Goulson, 2015) found that farms with pollen- and nectar-rich margins and/or floristically enriched buffer strips had a significantly positive effect on Bombus hortorum and Bombus lapidarius bumblebee nest density and foraging bumblebee abundance, but had no significant effect on B terrestris and B pascuorum, compared to farms with simple grass margins.

<table>
<thead>
<tr>
<th>Box 2-21 Evidence of the importance of field margins for pollinators</th>
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**Wildflower strips**

A literature review (Haaland, Naisbit and Bersier, 2011) found that in a significant majority of European studies, sown pollen- and nectar-rich flower mixtures and other wildflower strips support higher insect abundances and diversity than in other margin types such as sown grass margins and natural regeneration. A meta-analysis of 71 European studies of species richness and/or abundance of insect pollinators that compared sites with small-scale habitat creation measures (including field margin strips) with conventionally managed control sites (Scheper et al, 2013) showed that small-scale habitat creation practices enhanced pollinator richness, but their effectiveness varied with the magnitude of increase in flowering plant cover resulting from the practices, farmland type, and landscape context. Local effects were more positive in structurally simple landscapes (1-20% semi-natural habitat) than in cleared (<1% semi-natural habitat) or complex (>20% semi-natural habitat) landscapes, presumably because cleared landscapes lack sources of pollinator colonists and complex landscapes already have available habitat. In a UK study, the sown margins had 10-fold to greater than 100-fold more rare bumblebee species per sample area than generalized conventional conservation measures (Pywell et al, 2012).

**Grassy field margins**

A UK experiment (Carvell et al, 2007) found that 6 m-wide margins of cereal fields sown with pollen and nectar flower mixture supported significantly more foraging bumblebee species and individuals than cropped, grassy or naturally regenerated field margins. An earlier UK study (Marshall, West and Kleijn, 2006) showed that bumblebee abundance in July and August was significantly higher on pollen and nectar margins compared with wildflower margins, mature grass margins and recently sown grass margins. Bumblebee abundance along grassy ditch margins in Sweden (Persson and Smith, 2013) in late season (but not early season) was positively related to total amount of herbaceous flowers, and to the presence of potential nesting habitats in grass-clover leys and semi-natural pastures in the surrounding landscape. Ditch borders in simple landscapes had on average only 16% of the flower density found in other border habitats, while those in complex landscapes had 78%.

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14 most of the studies from the UK, a third from Switzerland, a fifth from Germany, the other studies from Sweden, Norway, Austria, Ireland, the Netherlands
**Common farmland birds**

The benefits for field margins for birds vary considerably depending on how the margins are established and managed. Whilst some species benefit from naturally regenerated or grass sown margins, these often develop dense and species poor vegetation on the highly fertile arable soils, with limited food resources and accessibility for birds. Cutting or scarification can increase their value for foraging birds, but they are likely to be of greatest value if sown with a bird seed mix. Research in the UK has showed that sympathetically managed field margins can provide a range of plant and invertebrate food resources for birds both in summer and winter at higher densities than in adjacent cereal fields (Vickery, Carter and Fuller, 2002) (see Box 2-22. In general, the best winter food supplies (mainly seeds) are provided by game cover / seed crops and naturally regenerated rotational strips. The most abundant summer food supplies (invertebrates and seeds) are provided by a diverse sward; grass/wildflower strips, uncropped wildlife strips and naturally regenerated rotational set-aside strips followed by conservation headlands (i.e. cereal headlands with reduced pesticide applications).

Species that tend to forage on field margins such as Yellowhammer (*Emberiza citrinella*) and Tree Sparrow (*Passer montanus*) are most able to benefit from these habitat features whilst whole-field approaches are required for boundary-avoiding species such as Skylark and Lapwing. However, a study in Sweden showed that cereal fields with grass buffer strips between field and ditch had more Skylark territories per ha up to 100m into the field, and higher invertebrate activity densities, compared to cereal fields that had no grass margins (Josefsson et al, 2013). Fields were matched to have identical management practices, field sizes and distance to landscape elements such as forest edge or semi-natural grassland.

**Box 2-22 Evidence of the importance of field margins to farmland birds in the UK**

**Sown versus grassy margins**

A study found that flower-rich sown margins had higher densities of plant flowers, buds and seeds suitable for bird food compared to grass-only margins (Westbury et al, 2017). Whilst grass margins managed by an annual cut had similar abundance of invertebrates suitable for bird food as species-rich sown margins, the birds’ access to the food resources is restricted by the sward density and lack of bare ground. Scarification of grass margins with a power harrow was highly effective at increasing invertebrate food availability but decreased flower and seed food resources (Westbury et al, 2017). A review of research found that Yellowhammer and Tree Sparrow used wildflower strips more than margins sown with grass seed only (Vickery, Feber and Fuller, 2009). Cutting grass margins significantly increased foraging Yellowhammers in late summer compared to uncut margins (Douglas, Vickery and Benton, 2009). Yellowhammers were more likely to locate territories in areas containing enhanced margins in the UK; i.e. where field margin habitats were sown with wild flowers and/or agricultural legumes (McHugh et al, 2016). One study concluded that sown legume field margins provided sufficient invertebrate biomass to support developing game bird chicks but may be too dense for foraging by game birds (Wood et al, 2013), whilst another found similar abundance of Yellowhammer chick food invertebrates in sown and grass margins but the more open sward structure of the sown margins increases prey accessibility and improves predator avoidance (McHugh et al, 2016).

**Species that are the focus of the Birds and Habitats Directives**

Some evidence was found that grassy field margins are foraging habitats for farmland species that are the focus of the Birds and Habitats Directives, such as Little Bustard (Lapiedra et al, 2011) and birds of prey that prey on small mammals that are abundant in grassy field margins (Rodríguez-Pastor et al, 2016).

**Box 2-23 Evidence of the importance of field margins for species that are focus of Birds and Habitats Directives**

**Little Bustard**

Grassy field margins in Mediterranean Spain were heavily used by foraging female Little Bustard after the breeding season in preference to grassland or crops (Lapiedra et al, 2011).

**Birds of prey in Spain**

In Spain (Rodríguez-Pastor et al, 2016), a survey of small mammals in arable fields found that grassy field margins were the most stable habitat, with densities about 2.3 times higher on average than within fields (and 8-9 times higher than in cereal fields in spring and autumn when soils are tilled and bare), and provide a key refuge in winter and spring before crops develop. These small mammals are the main prey items of a number of birds of prey on Spanish arable farmland that are the focus of the Birds Directive Annex II, such as Black-winged Kite and Hen Harrier.
2.4.3.5 Ponds

Ponds must have a maximum size of 0.1 ha according to Article 45. Member States can define a minimum size; can decide to include a strip of riparian vegetation alongside the pond up to 10m in width to count towards the size of the pond; and can establish criteria to ensure ponds are of natural value ‘taking into account the role that natural ponds play for the conservation of habitats and species’. Concrete or plastic reservoirs cannot count towards an EFA. For example, Hungary has included up to 10m of riparian vegetation. Belgium-Flanders and Italy set a minimum size of 0.01 ha including riparian vegetation. Germany has chosen to also include wetlands and biotopes as defined by national or regional legislation.

There is evidence that large numbers of farmland ponds have been lost particularly in Western Europe in recent decades (see Box 2-24).

Box 2-24 Evidence of loss of farmland ponds in Europe

Spain: Over two decades, a Mediterranean farmland landscape in Spain lost 56% of its temporary ponds, of which 89.3% were destroyed through agricultural activities such as cultivation, conversion to permanent farm ponds, and drainage (Ferreira and Beja, 2013). Mediterranean temporary ponds are a Natura 2000 priority habitat.

France: Over three decades (1975-2006), 57% of 199 ponds in the département Pas-de-Calais in north-western France were lost (Curado, Hartel and Arntzen, 2011). Pond disappearance appeared to be associated with a decrease in surrounding grassland and an increase in surrounding arable land. Small, artificial ponds with either natural substrate or concrete drinking troughs were more often lost than larger, semi-natural ponds.

Ireland: Farmland ponds declined by 54% between 1887-1913 and 2005-11. Most ponds and greatest losses were in the East, which coincides with agricultural intensification and human habitation (Reid et al, 2014). However, habitat loss for some species may have been partly compensated for by the expansion of artificial field margin ditches associated with drainage projects.

Wild native plants

Ponds with natural substrates and a riparian vegetation zone provide a habitat for a high plant richness compared to arable farmland. A study comparing the biodiversity of water bodies in three biogeographic regions of Europe found that the highest gamma diversity of macrophytes was associated with ponds and lakes (Davies et al, 2008), though alpha diversity was highest in rivers. Ponds also generally had the greatest numbers of unique species and the highest Species Rarity Index scores (the latter were only available for the UK). A study of 71 ponds in semi-natural habitats in England and Wales found that temporary ponds supported an average of 17 wetland plant species each (Nicolet et al, 2004).

Invertebrates

Ponds on farmland generally provide a suitable habitat for a high diversity of aquatic macro-invertebrates, over a range of sizes and substrates (see Box 2-25 for evidence).

Box 2-25 Significance of ponds on farmland for aquatic macro-invertebrates

A study of 120 ponds from four biogeographic regions in Europe found highest diversity in the Mediterranean and Atlantic regions (160 and 145 taxa respectively) (Céréghino et al, 2012). However these data may not relate solely to ponds on farmland, and the authors note that there is a lack of readily available data on European pond macroinvertebrates. A study of 36 artificial farmland ponds in southwest France found 114 macroinvertebrate taxa, of which 84% were uncommon in ponds and rivers, indicating that farmland ponds can make a significant contribution to macroinvertebrate biodiversity (Céréghino et al, 2008).

In a study of 94 artificial farm ponds in southern Spain with both natural and artificial substrata, insects were the most dominant group with 53 out of a total of 68 of the identified families (Fuentes-Rodríguez et al, 2013). The main families were Diptera, Coleoptera and Heteroptera.

Several large brachiopod crustaceans classified as vulnerable on the Portugal red list are largely restricted to temporary farmland ponds in the Mediterranean region (Sala et al, 2008).

Water beetles have been shown to be a suitable surrogate for overall pond invertebrate density (Bilton et al, 2006), and an Irish study of 54 farmland ponds found that over 30% of the national water beetle species fauna was present in the ponds (Gioria et al, 2010).
**Common farmland birds**

Bird species richness is higher on farmland that contains ponds or other water bodies, and some water birds have found breeding habitat in artificial irrigation ponds in response to loss of natural wetland breeding habitat (see Box 2-26).

<table>
<thead>
<tr>
<th>Box 2-26 Significance of ponds on farmland for birds</th>
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<tbody>
<tr>
<td>In a study in England comparing managed and unmanaged farmland ponds (diameter &lt;20km), it was found that avian species richness, abundance and bird-visit frequencies were all higher at managed open farmland ponds (Davies et al, 2016).</td>
</tr>
<tr>
<td>A study of 219 artificial irrigation ponds in a semi-arid region of southeastern Spain found that they were used as breeding and foraging habitat by 22 waterbird species and several species appeared to breed preferentially at these sites than in nearby natural and semi-natural wetlands (Sánchez-Zapata et al, 2005). Of the 30 species that breed in the area, 7 were observed to breed at these sites. The most abundant species were black-winged stilts (Himantopus himantopus), little grebes (Tachybaptus ruficollis) and little-ringed plover (Charadrius dubius). Coots, mallards and moorhens were found in lower numbers than in nearby wetlands.</td>
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**Species that are the focus of the Birds and Habitats Directives**

Farmland ponds are critically important habitats for many amphibians (see Box 2-27). Temporary (seasonal) ponds are critically important habitats for amphibians in Mediterranean landscapes, as they allow breeding in the absence of fish predators. However, many have been lost in recent times. Farmland ponds are key breeding habitats for protected amphibian species such as Great Crested Newt (Triturus cristatus). The loss of small ponds, temporary ponds and traditional artificial habitats such as wells, stone troughs, and irrigation channels in agricultural areas, has been assessed by IUCN experts as a major pressure on amphibian populations (Temple and Cox, 2009).

In contrast, the Common Frog *Rana temporaria* (not protected by the Habitats Directive) is not suffering the same decline in populations across Europe as other amphibian species (Temple and Cox, 2009). This may be because of the expansion of artificial field margin ditches associated with drainage projects, providing suitable replacement (Reid et al, 2014).

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<tr>
<th>Box 2-27 Significance of ponds on farmland for amphibians</th>
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<tbody>
<tr>
<td>Small cattle ponds in France are regularly used as amphibian breeding grounds, and their large-scale loss puts local populations at risk (Curado, Hartel and Arntz, 2011); the authors of this study recommend that conservation efforts focus on marsh and dune areas, and preserving ponds in the remaining grasslands.</td>
</tr>
</tbody>
</table>

**Protected species**

Temporary (seasonal) ponds created in arable farmland (less than 0.5 ha and 1 m deep) in northeast Spain (Ruhí et al, 2012) had similar or higher amphibian species diversity compared to natural local ponds after only two years, and were rapidly colonised by Natterjack Toad (*Bufo calamita*). A study of 57 Mediterranean temporary ponds in southwest Portugal found that rarer amphibian species, including Marbled Newt (*Triturus marmoratus*), occurred in the most species-rich ponds, which had longer hydroperiods, but permanent reservoirs also had lower abundances (Beja and Alcazar, 2003). However, Western Spadefoot Toad (*Pelobates cultripes*) occurred in all but the most permanent ponds. In general, these results reflect the conversion of ponds into irrigation reservoirs, with negative impacts on most species. In a study of ponds in a traditional rural landscape in Romania, Yellow-bellied Toad (*Bombina variegata*) was frequently present in the surveyed temporary ponds of sizes up to 0.02 ha (Hartel and von Wehrden, 2013), but was least common in overgrown ponds.

**Non-protected species**

Although farm pond numbers have declined significantly in Ireland, numbers of Common Frog (*Rana temporaria*) do not appear to have been affected as they are using other farmland habitats primarily drainage ditches (Reid et al, 2014). In the study in Portugal, Parsley Frog (*Pelodytes punctatus*) was generally confined to highly ephemeral ponds (Beja and Alcazar, 2003). Common Toad (*Bufo bufo*) was found in all the fishponds (sizes 0.2-5 ha, exceeding the EPA maximum) surveyed in (Hartel and von Wehrden, 2013) but rarely in the smaller temporary ponds.

### 2.4.3.6 Ditches

Ditches must have a maximum width of 6m according to Article 45. Open watercourses for the purpose of irrigation or drainage count, but channels with walls made of concrete are not eligible. As
examples of Member State rules, Italy set a maximum width of 10 m and Hungary has chosen to also include sweep wells. Ditches can provide valuable wet vegetated non-cropped habitats to both aquatic and terrestrial taxa, supply food resources lacking in otherwise dry and intensively managed cropland, and perform connectivity functions within a wider landscape (Herzon and Helenius, 2008). See also the evidence for biodiversity impacts of buffer strips along ditches. Drainage ditches are particularly a feature of lowland, moorland, and coastal areas. Ditch banks in Estonia were found to contain species typical of moist habitats and semi-natural grasslands (Aavik et al, 2008). The value of ditches for biodiversity depends on the type of management they receive, including frequency of cutting of the vegetation on the banks, dredging, and management to minimise pollution of ditch water with fertiliser, manure or pesticides. Some studies find that ditch communities have high biodiversity values; these tend to be ditches with continuity with wetlands and low pollution levels (Davies et al, 2008). In contrast, those associated with intense agricultural activities and hence increased runoff may have lower biodiversity, as high levels of herbicides, pesticides and fertilisers may impact floral and thus faunal richness.

**Wild native plants**
Ditch banks can provide habitat for a high level of wild plant species richness and for rare and specialised species, but their value for plant diversity is very dependent on the management. Ditch banks managed under agri-environment schemes generally have higher plant species richness compared to banks managed outside scheme requirements (Dicks et al, 2013), but some schemes fail to increase plant species richness as the management is not sufficient to reduce soil fertility from nutrient inputs from the arable land sufficiently to allow plants adapted to lower nutrient levels to colonise and maintain their abundance (Blomqvist, Tamis and de Snoo, 2009). Ditch banks that are not exposed to herbicide and inorganic fertiliser use in the arable field are higher in plant species richness (Manhoudt, Visser and de Snoo, 2007). Ditch banks on intensive dairy farms managed under agri-environment payments, and concluded that the late mowing is not sufficiently reducing the impacts of nutrient input from the farm on plant productivity (Blomqvist, Tamis and de Snoo, 2009). Another Dutch study found that ditch banks had highest plant species richness when adjacent to organic or “ecologically managed” fields with pesticide and nutrient free buffer zones (Manhoudt, Visser and de Snoo, 2007). A comparative study of waterbodies in three biogeographic zones in Europe found that ditches had the lowest macrophyte alpha diversity compared to other waterbodies, but their gamma diversity was not greatly different from that of rivers and streams, though inferior to that of ponds (Davies et al, 2008).

**Box 2-28 Significance of ditches for wild native plants**
Five out of eight comparative studies from the Netherlands and the UK in a systematic review (Dicks et al, 2013) found that managing ditches resulted in increased plant species richness and emergent plant cover in ditches managed under agri-environment schemes, whilst the other studies found no clear effect on biodiversity. One study in the Netherlands found no improvement of plant biodiversity over 10 years on ditch banks on intensive dairy farms managed under agri-environment payments, and concluded that the late mowing is not sufficiently reducing the impacts of nutrient input from the farm on plant productivity (Blomqvist, Tamis and de Snoo, 2009). Another Dutch study found that ditch banks had highest plant species richness when adjacent to organic or “ecologically managed” fields with pesticide and nutrient free buffer zones (Manhoudt, Visser and de Snoo, 2007). A comparative study of waterbodies in three biogeographic zones in Europe found that ditches had the lowest macrophyte alpha diversity compared to other waterbodies, but their gamma diversity was not greatly different from that of rivers and streams, though inferior to that of ponds (Davies et al, 2008).

**Invertebrates**
Ditches may provide habitats for rare and unusual aquatic invertebrates not found in other farmland habitats, such as species typical of temporary water bodies, as many ditches only hold water seasonally (see Box 2-29 for evidence from the UK).

**Box 2-29 Significance of ditches for aquatic invertebrate biodiversity in the UK**
A survey of the aquatic macroinvertebrate diversity (alpha, beta and gamma) of agricultural drainage ditches in a predominantly arable region in the UK (Hill et al, 2016) found high spatial and temporal differences among ditches, indicating that main arterial and side ditches make unique contributions to aquatic biodiversity. An earlier comparative survey of small waterbodies in England found that ditches (most of which were seasonal) were the least species-rich habitat compared to rivers and ponds, but supported uncommon species, including temporary water invertebrates not recorded in other waterbody types (Williams et al, 2004).

Many invertebrate species that are predators of crop pests have larvae associated with ditches, and a study found that the proximity of watercourses enhanced the abundance of adult predatory Daggerflies (Empididae) in borders of arable fields in Germany and of adult predatory Long-legged Flies (Dolichopodidae) in borders of arable fields in Italy (Pfister et al, 2017).
**Common farmland birds**

Certain farmland bird species can benefit from the presence of ditches on farmland, provided the ditches and banks are managed in suitable ways to provide foraging and breeding habitat (see Box 2-30 for evidence).

**Box 2-30 Significance of ditches for common farmland birds**

A survey in the Baltic countries (Marja and Herzon, 2012) found species richness and abundance of farmland birds were positively associated with the presence of ditches with tall vegetation, but did not find a clear impact of ditches with only grassy margins. Crop and grassland fields with open drainage ditches in Finland (Marja et al, 2013) were found to provide a higher habitat value for nesting Meadow Pipit (*Anthus campestris*) than fields with subsurface drainage. Birds closely associated with drainage ditches are Reed Warblers (*Acrocephalus scirpaceus*, *A. palustris*, *A. schoenobaenus*) and Reed Bunting (*Emberiza schoeniclus*), all of which need persistent reed stands and tall emergent plants (Herzon and Helenius, 2008).

Five out of eight comparative studies from the Netherlands and the UK in a systematic review (Dicks et al, 2013) found that managing ditches resulted in increased bird visit rates and higher numbers of some bird species or positive impacts on some birds in plots with ditches managed under agri-environment schemes, whilst the other studies found no clear effect on biodiversity.

**Species that are the focus of the Birds and Habitats Directives**

Ditches surrounded by intensive agriculture tend to be managed with low water levels and subject to regular vegetation management, which means that their conservation value is lower than other waterbodies. However, even drainage ditches in intensive arable farmland occasionally support species of conservation interest (Mountford and Arnold, 2006).

### 2.4.3.7 Traditional stone walls

The height and width of EFA stone walls must be defined by the Member States based on national or regional characteristics. For example, France defined 0.5-2m height, 0.1-2m width; Italy defined 10m minimum length, 0.3-5m height, 0.5-5m width, UK-Northern Ireland\(^\text{15}\) defined 0.5-2.3m height, 0.25-4m width, 3m minimum length; UK-Wales defined 1m minimum height, 4m maximum width, 20m minimum length. UK-Wales specify that stone walls with a hedge on top must be classified as walls rather than hedges. Germany\(^\text{16}\) also includes stone heaps and stone fences.

Stone walls are field boundaries characterised by layers of stones with spaces in between often covered with grasses and herbs and sometimes with scrub and woody species. They provide a variety of microhabitats including warm dry shelter and shade within the stones, and sunny dry exposed stone surfaces on the sides and top of the wall, which remain undisturbed for long periods of time. Stone walls are the most common farmland feature in southern Sweden where the density of stone walls is greater than 1km of stone wall per 100ha (Farmer et al, 2008). They are also abundant on arable farmland in many Mediterranean countries, and in traditional mixed farming landscapes of eastern Europe, for example Romania. A review in Ireland (Collier, 2013) found evidence that they function as platforms for supporting endemic flora and fauna, refuges for disparate species, linkages or corridors, seed reserves, shelter for scrub and other adjacent habitats, and nesting and roosting locations in exposed areas.

**Wild native plants**

Stone walls often have rich communities of lichens, mosses and ferns as well as many other plants associated with them.

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\(^{16}\) http://www.lei-web.de/app/greening/erlaeuterungen.pdf
**Invertebrates**

Dry stone walls will offer many micro-habitats and refuges for spiders, beetles, and other invertebrates that are natural enemies of arable crop pests.

**Common farmland birds**

Wheatears (*Oenanthe oenanthe*) commonly nest in the crevices of dry stone walls, and a study in Wales, UK, found their distribution was closely associated with length of dry stone wall (Woodhouse et al, 2005). However, this study was on permanent grassland farmland rather than arable, so is of limited relevance to EFA impacts.

**Species that are the focus of the Birds and Habitats Directives**

Stone walls are favoured habitats for reptiles, such as Slow-Worm (*Anguis fragilis*), snakes, and amphibians. Reptiles find stone walls provide favourable habitats, as the environment surrounding stone walls is often dry, bright and warm, and the stones store heat, whilst the inside of the wall and the vegetation around the base provide shelter and shady retreats to cool down and rest. A study of stone walls in a mountain area of the northern Appennines (NW-Italy) found they were a key habitat for the salamander *Hydromantes strinatii* and for a range of mollusc species (Manenti, 2014).

### 2.4.4 Buffer strips

Buffer strips eligible to count towards an EFA are strips covered by permanent grassland provided these are distinct from the adjacent eligible agricultural area. Member States must include those buffer strips along water courses protected under GAEC1, SMR1 or SMR10, and can also include other buffer strips as set out in Article 45 of Commission Delegated Regulation (EU) 639/2014. The Article 45 rules state that:

- The minimum width of ‘other’ buffer strips can be set by the Member State, but must not be below 1 m;
- There must be no production on the buffer strip, although grazing or cutting is permitted, provided that the buffer strip remains distinguishable from adjacent agricultural land;
- The strips must be located on or adjacent to an arable field, with the long edge parallel to the edge of a water course/water body; and
- Where they are along water courses, they can include strips of riparian vegetation up to a maximum width of 10 m.
- Member States can allow buffer strips that are not protected by cross-compliance to be prepared for agricultural use from 1 August if the area is to be used for harvest in the next year (e.g. Germany) (Hart, 2015).

Member States can also choose to allow an equivalent practice: ‘Management of uncultivated buffer strips and field margins (cutting regime, local or specified grass varieties and/or seeding regime, reseeding with regional varieties, no use of pesticides, no disposal of manure and/or mineral fertilisers, no irrigation, no soil sealing’.

Depending on Member States rules under the Water Framework Directive and cross-compliance, buffer strips can include:

- Permanent grassed strips
- Strips sown with annual seed mixes
- Bare soil strips
Literature reviews on the effects of farming practices associated with the CAP greening measures on climate and the environment

**Wild native plants**
The requirement that buffer strips should not be treated with fertiliser and for vegetation to be cut and removed to prevent eutrophication helps to maintain a higher level of plant diversity than typical field margins on arable farmland. However, plant species richness will vary greatly according to width of the buffer and management (see Box for evidence).

**Box 2-31 Evidence of impacts of buffer strips on wild plant diversity**
Watercourse buffer strips were compared with grass field margins managed for biodiversity in three English catchments (Critchley et al, 2013). The buffer strips had high structural diversity and bumblebee foodplant richness, but lower botanical value, diversity of foodplants for farmland birds and butterfly larvae, and perennial forbs important for invertebrates than the grass margins. Buffers established by species-rich seed mixtures or natural regeneration were superior to those established with a simple grass seed mixture. A study in the Netherlands found that sown species rich field margins sown next to buffer strips significantly increased the plant species richness of the buffer strips over four years, associated with a fall in soil nitrate concentrations (Musters et al, 2009). Two year old sown grass buffers next to cereal fields in France did not have particularly high plant or invertebrate biodiversity compared to fields and hedgerows, but late mown buffer strips next to maize, which is harvested late, were found to have flowers likely to be visited by hoverflies, in particular yellow flowers such as *Lotus corniculatus* (Fabaceae), *Sonchus asper* (Asteraceae) or *Ranunculus* spp. (Ranunculaceae) (Ernoult et al, 2013).

**Invertebrates**
Wide, floristically diverse buffer strips provide suitable habitat for a range of invertebrates, including pollinators, natural enemies of crop pests and soil macro-invertebrates (see Box 2-32 for evidence). Grass buffer strips across arable fields can also provide refuges for invertebrate natural enemies of crop pests (see Box 2-19 for evidence). The management of buffer strips influences their benefits for biodiversity, as shown by a UK study which found that buffer strips under agri-environment increased micromoth abundance, though not species richness, compared to buffer strips not under a management scheme (Fuentes-Montemayor, Goulson and Park, 2011). See field margins section for further evidence.

**Box 2-32 Evidence of impact of buffer strips on invertebrates**
Three reviews and 16 comparative studies from Finland, the Netherlands, Sweden and the UK in a systematic review (Dicks et al, 2013) found that planting wide grass buffer strips (some of the margins were floristically-enhanced) increased arthropod abundance, species richness and diversity (spiders, sawfly larvae, butterflies, grasshoppers, beetles, bees). For example, a field-scale study in southern England found that plants, bees (Apidae) and grasshoppers (Orthoptera) were all more abundant or had higher diversity on arable fields with 6 m-wide sown grassy margins, compared to control fields without margins (Marshall, West and Kleijn, 2006). Six of the reviewed studies from the Netherlands and the UK found that planting grass buffer strips had no clear effect on insect numbers.

A study found that the distribution of a rare bumblebee *B. muscorum* in central Germany was closely associated with ditch margins in open arable farmland (Diekötter et al, 2006). Because *B. muscorum* exhibits small foraging ranges, it is essential that landscape elements that provide nesting sites, foraging habitats and undisturbed hibernation structures are next to each other, and the study indicated that the supply of these habitat elements may have reached a critically low threshold in the study region.

**Common farmland birds**
See field margins for evidence.

**Species that are the focus of the Birds and Habitats Directives**
No specific evidence found.

2.4.5 **Agroforestry**
To be eligible to contribute to an EFA, areas of agroforestry must currently receive support from the CAP under the forestry measures or have received support in the 2007-2013 period, and must continue to comply with the conditions set out for receipt of RDP funding. Agroforestry must be established on agricultural land, either grassland or arable. Christmas trees and fast-growing species for short-term cultivation are excluded from support. Member States were required to define
maximum densities of plantation of forest trees. The agroforestry land counting towards an EFA can be located anywhere on a farm holding unless the Member State has specified otherwise.

Agroforestry can be silvo-arable, i.e. trees with crops, or silvo-pastoral, i.e. trees with grassland that is grazed and/or mown. Silvo-pastoral systems are much more common in Europe, including traditional orchards, grazed dehesa or montado, wooded pastures, and parklands (Rigueiro-Rodríguez, McAdam and Mosquera-Losada, 2009). The silvoarable systems of Europe can be split into those typical of northern Europe and those of the Mediterranean, which cover a greater area and include a greater diversity of systems due to the broader range of commercial tree and crop species grown (Eichhorn et al, 2006). In general, the systems of northern Europe are limited by light, whilst those of the Mediterranean are limited by the availability of water.

There is relatively little evidence of the impacts of silvo-arable agroforestry systems in Europe on biodiversity. A meta-analysis of 53 publications on European agroforestry systems found no significant effects of silvo-arable agroforestry on biodiversity, but silvo-pastoral agroforestry did have a significant positive effect on biodiversity compared to specialised agricultural and forestry systems (Torralba et al, 2016).

**Wild native plants**
Silvoarable forestry is expected to increase landscape diversity, which can provide a greater variety of micro-habitats for plants (Palma et al, 2007b).

**Invertebrates**
In theory, agroforestry helps reduce pest problems because tree-crop combinations provide greater niche diversity and complexity than monoculture systems of annual crops. The mechanisms that might be operating in arable agroforestry include: wide spacing of crop rows may make them more difficult to find by pests; trees or associated plants may serve as a trap-crop to detour pests away from the crop, or they may be repellent to the pests; trees may attract more predators and parasites than monocultures, for example by providing nectar and pollen, thus reducing pest density through predation; the windbreak effect of trees may enhance the activity of parasitoids and predators. Silvopastoral agroforestry in the UK was found to increase carabid beetle and spider abundance and diversity which attracted more birds (McAdam et al, 2007).

**Common farmland birds**
See short rotation coppice for evidence of impacts of tree planting on birds.

**Species that are the focus of the Birds and Habitats Directives**
No evidence found.

**2.4.6 STRIPS OF ELIGIBLE HECTARES ALONG FOREST EDGES**
Member States can choose whether to allow strips of land between 1m and 10m wide along forest edges with agricultural production and/or without. Strips without production can be grazed or cut as long as the strip remains distinguishable from the adjacent arable land.

There is evidence that structurally complex forest edges are important refuges for biodiversity in agricultural landscapes (Marshall and Moonen, 2002). Low-contrast (‘soft’) edges are wider than high-contrast (‘hard’) edges and provide an extension of the ecotone habitat for both open-habitat specialists and native forest species. In contrast, abrupt forest edges along forest stands with high tree density provide no ecotone habitat between the highly shaded closed forest and the open field, whilst the tree shade reduces crop production at the field edge. However, no specific evidence was identified of the impact of introducing a field margin on arable fields bordering forests independently of changes in the structure of the forest edge itself.
Wild native plants
Permanent arable field margins along forest edges provide plant habitats that differ ecologically from field margins along low (frequently cut) hedges or along non-woody boundaries, as they are more strongly influenced by the shading and shelter provided by the forest trees, favouring plant species that are tolerant of shade and humidity. Infrequently disturbed margins can provide habitat for plant species that colonise from the forest. No specific evidence for this was found in the literature.

Invertebrates
A study in France found that species assemblages of ground beetles within 15m of the forest edge were a mix between forest and arable field species assemblages, and that ground beetles active in the crop fields overwintered in the first few metres of forest edges, with a highly spatially structured distribution as species showed different preferences for forest edge orientation (Roume et al, 2011). Authors note that the presence of ground beetles belonging to forest species in crop fields could be beneficial to agriculture because many of them are known to prey on pests such as slugs. However, the field edges in the study were abrupt with only narrow field margins, and it is not possible to draw conclusions from this study about the possible impact of wider field margins on invertebrate predator diversity. A study in southern France showed that the aphid predatory hoverfly *Episyrphus balteatus* overwinters in larval, pupal and adult stages in forest edges, from which it spreads into crop fields in spring, and in summer abundance was correlated with flower abundance in forest edges (Sarthou et al, 2005).

Forest edges adjacent to arable land can provide suitable nesting sites and pollen and nectar resources for foraging bees and hoverflies (Garibaldi et al, 2014). A scheme in the Netherlands that funded the cutting of a strip of the forest edge with subsequent natural regeneration led to a significant increase on butterfly numbers five years after intervention, primarily due to the increased abundance of nectar-providing flowers in the forest edge strip (Non and de Vries, 2013). However, the strip was established within the forest rather than the field.

Common farmland birds
Some common farmland and forest birds find favourable habitat in complexly structured forest edges, whilst specialist open land bird species avoid forest edges (see Box 2-33). A forest edge margin can have the effect of reducing predation risk on ground-nesting birds by reducing the contrast in height and structure between crop and non-crop, compared to a field with no margin between forest and crop (see Box 2-33).

> **Box 2-33 Evidence of impacts of forest edges on common farmland birds**
> A study assessed bird abundances at varying distances from the forest edge in landscapes with variable amounts and spatial configurations of forest plantations in southern Portugal, and found that overall bird density of farmland and ground-nesting birds declined with distance away from forest edges (Reino et al, 2009). The authors concluded that many farmland birds find better breeding conditions in the relatively tall grassland strip along forest edges, characterised by small shrubs, than in the regularly grazed fields. A Swedish study showed that the risk of predation of Northern Wheatear (*Oenanthe oenanthe*) nests in crop field edges was higher when the forest edge bordered directly onto the crop field without a vegetated margin, i.e. the contrast between tall and short vegetation was abrupt, but predation was lower where the field margin vegetation was similar in height to the crop (Schneider et al, 2012). This edge effect on nest predation risk was evident during the incubation stage but not the nestling feeding stage.

Species that are the focus of the Birds and Habitats Directives
Several forest bird species with unfavourable conservation status in France (e.g. hoopoe *Upupa epops*, mistle thrush *Turdus viscivorus*) showed positive responses to the density of graded native broadleaved forest edges adjacent to arable fields in mosaic landscapes in France, whereas there were no positive bird responses to the abrupt edges of pine forestry plantations lacking understorey vegetation (Terraube et al, 2016).
2.4.7 Short rotation coppice

Short-rotation coppice (SRC) or short-rotation woody cropping (SRWC) refers to silvicultural systems designed to produce woody biomass using short harvest cycles (1–15 years), intensive silvicultural techniques, high-yielding varieties, and often coppice regeneration. The SRC species most frequently mentioned in the literature are also the most popular species chosen by those Member States which allowed SRC to count towards an EFA, namely willow (*Salix*), poplar (*Populus*), alder (*Alnus*), birch (*Betula*) and ash (*Fraxinus*) (SWD(2016) 218 final). Two non-indigenous species have been allowed on EFA - Eastern Black Walnut (*Juglans nigra*) in Hungary and Black Locust (*Robinia pseudoacacia*) in Romania. The use of *Robinia* spp in Romania could be a concern, given that it is a non-native species and can be very invasive in open habitats (Hart, 2015). It is a common tree used for shelterbelts and plantations in Romania and recommended for the restoration of degraded soils since it grows quickly, fixes nitrogen and improves soil organic matter. However, it can be problematic if planted on sandy grasslands (Vítková et al, 2017). *Robinia* has been reported as a threat to a number of Natura 2000 habitats and controlling *Robinia* plantations is mentioned as a management measure in several Natura 2000 sites.

A meta-analysis of the comparative effects on biodiversity of SRCs relative to a range of cultivated and uncultivated habitat types found that, in general, willow and poplar SRC crops were assessed as better for biodiversity than arable land for almost all taxa (birds, mammals, butterflies, canopy invertebrates, ground beetles, rove beetles, spiders, earthworms, other soil fauna, plants) (Dauber, Jones and Stout, 2010). SRC biodiversity benefits were equal to or lower than woodland habitats. Studies comparing SRCs to set-aside and grassland found mixed results. Results also varied depending upon SRC type. As the trees remain in situ for at least 5 up to 20 years, the soil is not cultivated annually and SRC provides a stable habitat that is only partially shaded over by the trees. However, the high tree densities in commercial SRC result in low light levels reaching the ground after five years of tree establishment (McKay, 2011), which significantly alters the ground flora and fauna. Short rotation coppice is expected to be more homogeneous in terms of tree species and age class than natural woodland and mature plantations, and most closely resemble young afforestation or single species coppice.

**Wild native plants**

A review of European studies concluded that SRC willow and poplar plantations increase farmland flora diversity compared with arable crops (Rowe, Street and Taylor, 2009). Plant species richness in SRCs was found to be lower than in set-aside fallow but higher than on arable land and forest plantations (Rowe et al, 2011), but depends on the harvest cycle intensity and varies over time (Baum, Bolte and Weih, 2012), and is mainly defined by the species pool of ruderal species in the soil seed bank and adjacent vegetation (Baum, Weih and Bolte, 2013). As the crop stand architecture of SRCs is highly dynamic, SRCs are suitable for a mixture of species typical of agricultural fields or grasslands, hedgerows, ruderal plots and forests, resulting in high plant species richness over time, but also a high turnover of species composition (Glemnitz et al, 2013). However, forest species with low colonisation ability, such as lichens and bryophytes, are unlikely to have time to develop in SRC because of the short rotation period (McKay, 2011). Several studies conclude that SRCs exhibit certain unique habitat conditions that distinguish their biotic communities from those of all other biotopes within the agrarian landscape (Glemnitz et al, 2013; Rowe et al, 2011), including occurrence of some flowering species that provide food resources

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17 Under Article 4(2)(c of Regulation (EU) 1307/2013 according to definition of short rotation coppice in Article 4(1)(k) of the same regulation
18 SFC Calimani-Gurghiu - Securing favourable conservation status for priority habitats from SCI Calimani-Gurghiu, LIFE08 NAT/RO/000502 (http://ec.europa.eu/environment/life/project/Projects/index.cfm?fuseaction=search.dspPage&n_proj_id=3546)
19 Sturm U and Wilke C (eds) (2012), Management practices for invasive species in Danube Delta Biosphere Reserve (Romania) and Triglav National Park (Slovenia) (http://www2.ioer.de/download/habit-change/HABIT-CHANGE_3_4_1_A_Invasive%20species%20management%20in%20DDBR%20and%20TNP_update.pdf)
for invertebrates such as bees and butterflies and a higher plant biomass than on arable land (Haughton et al, 2015). The authors of a German study conclude that short-rotation coppice strips contribute little to rare-species conservation, but have substantial effects on habitat connectivity and ecosystem services, as the herbaceous ground vegetation in SRCs developed similarities with grassland or field margin vegetation, a vegetation type which is very scarce in many agricultural landscapes in Germany (Glemnitz et al, 2013). A review concluded that the development of ground vegetation within SRC plantations is of great importance for the richness of the associated invertebrate communities and for the food availability and shelter for birds, but commercial SRCs managed for maximum biomass yield are subject to intensive weed control with herbicides and/or mechanical treatment (Dauber, Jones and Stout, 2010).

**Box 2-34 Evidence of impacts of short rotation coppice on wild native plant biodiversity**

A review of European studies of the impacts of SRC willow and poplar on plant diversity in Europe concluded that SRC plantations can increase farmland flora diversity compared with arable crops (Rowe, Street and Taylor, 2009). Another review of 47 studies of the impacts of SRC crops found clear positive effects on plant species richness compared to arable land but ambiguous results in the few studies that compared species richness of biomass crops with grasslands (Dauber, Jones and Stout, 2010). Short-rotation coppice plantations of willow (Salix spp) and poplar (Populus spp) were richer in plant species (mean: 30 species per 100 m²) than arable land (10 species), coniferous forests (13 species) and mixed forests in Germany, but their value for diversifying plant species richness at the landscape-scale depends on harvest cycle intensity and varies over time (Baum, Bolte and Weih, 2012). As the soil seed banks of German and Swedish SRC plantations (Baum, Weih and Bolte, 2013) contained predominately ruderal species with almost complete absence of woodland species, authors conclude that the phytodiversity in SRC plantations is mainly due to the species pool of the adjacent vegetation and site conditions. In a UK study, weed biomass was found to be consistently greater in short rotation coppice willow than in cereals, and the long-flowering Epilobium spp. dominated (Haughton et al, 2015). Another UK study found that mature (at least 5 year old) willow SRC supported a different plant community to neighbouring set-aside fallow and arable land, being dominated by competitive perennial species such as Elytrigia repens and Urtica dioica, although plant species richness was lower than in set-aside fallow (Rowe et al, 2011). A German study of plant communities in SRCS (Glemnitz et al, 2013) concluded that SRCS exhibit certain unique habitat conditions that distinguished their biotic communities from those of all other biotopes within the agrarian landscape. As the crop stand architecture is highly dynamic, the SRCS are suitable for a mixture of species typical of agricultural fields or grasslands, hedgerows, ruderal plots and forests and resulting in high species richness. Herbaceous ground vegetation developed toward the grassland or field margin vegetation, a vegetation type which is really scarce in many agricultural landscapes in Germany. The authors conclude that short-rotation coppice strips contribute little to rare-species conservation, but have substantial effects on habitat connectivity and ecosystem services.

**Invertebrates**

SRCS appear to have mixed impacts on invertebrates (see Box 2-35 for evidence). The impacts depend upon the taxa compared, as well as on the land type replaced and on the SRC management practices. For example, applications of fertiliser or compost or mulching with plastic or plant residues are expected to significantly influence the soil macrofauna (Baum et al, 2009).

Short rotation coppice and the absence of tillage results in leaf litter accumulation on the soil surface, with higher C/N ratio and lignin content than crop residues (Baum et al, 2009), and this has been shown to increase the abundance of earthworms and woodlice compared to arable soils (Makeschin, 1994 cited in Baum et al, 2009; Haughton et al, 2015), although earthworms were not more abundant than under grassland (Dauber, Jones and Stout, 2010). The colonisation of willow and poplar roots by ectomycorrhizal fungi drives changes in the soil microbial activity and therefore also the soil macrofauna (Baum et al, 2009). As SRC rotations are typically 8-20 years in length there is not enough time for substantial dead wood resources to develop and therefore the ability of this habitat to support saproxylic species is expected to be low (McKay, 2011), but densities of saproxylic forest arthropods have been shown to increase as SRC ages (Glemnitz et al, 2013).

Various studies have found that the abundance of invertebrate predators is greater in SRC compared to arable crops (see Box 2-35 for evidence), including a large scale assessment in commercially sized fields across the UK (Haughton et al, 2015). However, the only identified study that directly compared predation rates of pests found no differences between willow SRC and cereal crops (Rowe et al, 2013).

There is evidence that foraging bees and butterflies benefit from SRCS with flowering willow (Salix spp) (Haughton et al, 2009; Rowe et al, 2011; studies cited in Rowe, Street and Taylor, 2009), but it is not possible from these studies to draw any conclusions about the value of SRC as breeding habitat for bees or butterflies.
Box 2-35 Evidence of impacts of Short Rotation Coppice (SRC) on invertebrates

Soil and detritivore invertebrates
SRC results in leaf litter accumulation on the soil surface in the absence of tillage, with higher C/N ratio and lignin content than crop residues (Baum et al, 2009). The non-tillage management and the high litter supply can change the abundance and diversity of the soil fauna. Willows and poplar can be colonised by ectomycorrhizal fungi, which changes soil microbial colonisation and activity, driving change in soil macrofauna compared to arable soils, although the nature of changes is not well understood (Baum et al, 2009). A German study of arthropod communities in SRCs (Glemnitz et al, 2013) found that with increasing shading by trees, the densities of forest arthropods that require wood structures increased. A study in Germany found increases in abundance and mass of earthworm, woodlice and harvestmen under poplar and willow SRC compared to the previous arable soil (Makechin, 1994 cited in Baum et al, 2009), whilst another study found no differences in species richness earthworms in SRC willow compared with grasslands (Dauber, Jones and Stout, 2010). A UK study found earthworm numbers decreased over 6 years on five SRC sites (Coates and Say, 1999, cited in Rowe, Street and Taylor, 2009).

Predatory invertebrates
A comprehensive, standardized assessment of commercial fields across the UK found that the densities of soil macro-invertebrate groups and natural enemies were many times greater in SRC willow compared with break crops commonly used in the arable rotations (Haughton et al, 2015). A review of 47 studies of the impacts of SRC crops on biodiversity in Europe or the US cited three studies that found lower species richness of ground beetles (Carabidae), and, in some of the studies, rove beetles (Staphylinidae) in SRC crops compared to arable land (Dauber, Jones and Stout, 2010). Two UK studies identified 30 species of ground beetle (Carabidae) and 15 species of rove beetles (Staphylinidae), and a maximum of 27 ground beetles and 25 rove beetles at any one site at five SRC sites, species numbers that are much higher than the average number of species of phytophagous invertebrate utilizing cereal crops (Sage and Tucker, 1998, Coates and Say, 1999, cited in Rowe, Street and Taylor, 2009), whilst in a German study carabid beetles (Carabidae) increased in species diversity but decreased in abundance in poplar and willow SRC compared to arable soils (Makechin, 1994 cited in Baum et al, 2009). A study in England found that willow SRC had higher abundance and diversity of ground-dwelling arthropod predators than did arable land and set-aside, but there was no difference in predation rate of fly pupae in willow SRC compared to cereal crops, and no differences in the combined activity-density of the two principal Coleoptera families (carabids and staphylinids) that predate fly pupae, as the increased abundance of Staphylinids compensated for the lower activity of Carabids in SRC (Rowe et al, 2013).

Lepidoptera
A study of SRC in the UK reported that the boundary of SRC willow contained both a higher foraging butterfly abundance and species richness than arable controls (Cunningham et al 2004, cited in Rowe, Street and Taylor, 2009). Of the 22 species recorded, none were found exclusively in the SRC, and all were relatively common and widespread species. Another UK study recorded 14 species of butterflies in SRC plantations of which most were common, migratory or colonial polyphagous species with weed or stress tolerant food plants (Sage et al 1994, cited in Rowe, Street and Taylor, 2009). Another UK study found that total butterfly abundance was greater in field margins of willow biomass crops than in arable field margins (Haughton et al, 2009). All these studies recorded only adult foraging butterflies, and therefore cannot be used to draw conclusions with regard to the suitability of SRC as butterfly breeding habitat.

Pollinators
A Danish study concluded that while ground flora in SRCs represents a poor source of nectar due to both the species present and the limited flowering under a mature crop canopy, flowering of the willow stools in the 2nd and 3rd years of growth may constitute an important early season source of nectar and pollen for flower visiting insects such as bees (Reddersen, 2001, cited in Rowe, Street and Taylor, 2009). A UK study (Rowe et al, 2011) found that the abundance of winged invertebrates was similar in mature willow SRC, arable crops and set-aside, but taxonomic composition varied greatly, and Hymenoptera (bees and wasps) and large Hemiptera (bugs) were more common in willow SRC than in the other habitats.

Common farmland birds
The evidence for the impacts of SRCs on common farmland birds suggests that these will depend upon the habitat preferences of the individual species investigated, and on the type of land SRCs are compared to (see Box 2-36 for evidence). A review of European studies concluded that SRC willow and poplar plantations increase the abundance of common birds associated with scrub and woodland habitats, such as Blackbird, Chaffinch, Dunnock, Great Tits, Reed Bunting, Willow Warbler, and Wren, compared with arable crops, and also some rarer species (Rowe, Street and Taylor, 2009). Another review concluded that the bird species composition of the SRC crops did not resemble forest bird communities but open farmland or transitional scrubland communities (Dauber, Jones and Stout, 2010). The literature indicates that SRC may be valuable for birds preferring woodland or edge habitat in locations lacking such habitats, but is likely to offer fewer benefits than mature or intensively managed woodland. Certain species with strong preference for open farmland, such as Skylark, are not favoured by SRCs.
Surveys of birds using willow short-rotation coppice (SRC) fields over 5 years in England (Sage, Cunningham and Boatman, 2006) found that most existing farmland bird species continued to use SRC and some species characteristic of scrub and woodland habitats were attracted onto farmland by SRC, whilst Yellow Wagtail and certain locally occurring farmland specialists may be displaced by SRC plantations, and Skylark used only cut SRC fields. The interior of large SRC plots contained fewer birds than the edge-zone (< 50 m). Overall, more individuals and species were recorded in and around SRC than equivalent arable or grassland throughout the year. A comparison of bird groups in miscanthus (Miscanthus x giganteus) fields, plots of short rotation coppice (SRC) willow, arable crops and grassland in two winters and one summer in southwest England (Sage et al, 2010) found more species and individuals in SRC than miscanthus during the summer. As stubble in May, the miscanthus contained broadly similar species at similar densities to arable and grassland comparison plots, but during the winter the miscanthus held more wood/scrub species such as Blackbirds, tits, Reed Buntings and Woodcock than arable and grassland plots. The authors suggest that bird use of miscanthus in summer and winter is likely to be variable, affected by region, weediness, crop structure and patchiness.

An inventory of farmland birds in 212 point-count sites in south-central Sweden found positive associations of bird occurrence of four common species with short rotation coppice (Reed Bunting, Pheasant, Linnet, Common Whitethroat) (Berg et al, 2015). A German study of bird communities in SRCs (Gleinitz et al, 2013) found that with increasing shading by trees, the densities of bird species that require wood structures increased. A meta-analysis of studies of the impacts on birds of conversion of existing, intensively managed poplar (Populus) forests to short-rotation woody cropping (Riffell et al, 2011) found that diversity and abundance of bird guilds were lower on short-rotation plantations compared with reference woodlands, while abundance of individual bird species was more variable and not consistently higher or lower on SRWC plantations. Shrub-associated birds were more abundant on SRWC plantations, but forest-associated and cavity-nesting birds were less abundant. However, the authors note that none of the reviewed studies directly compared SRWC with intensively managed forests, so the findings are tentative. Plantations of SRWCs could contribute to overall landscape diversity in forest-dominated landscapes by providing shrubby habitat structure for nonforest species. However, extensive conversion of mature or intensively managed forests to SRWC would likely decrease overall diversity, especially if they replace habitat types of high conservation value.

Species that are the focus of the Birds and Habitats Directives

The rapid growth rate of SRC trees is likely negatively affect a few bird species which require open farmland, such as Yellow Wagtail, Grey Partridge and Stone Curlew where they are present on arable farmland (Rowe, Street and Taylor, 2009).

SRC is likely to be browsed by large mammals such as deer or moose where they are present on farmland (Rowe, Street and Taylor, 2009).

2.4.8 AFFORESTED AREAS

Afforested land qualifies as EFA if it was afforested using rural development funding or compliant national schemes since 2001. The age of EFA afforested stands therefore ranges from minimum 2 to maximum 15 years old, at which age most forest trees will not yet have reached their full canopy size, and will be a quarter of the way or less to reaching maximum timber potential.

Up to 2013 afforestation was funded by rural development on either agricultural land or non-agricultural land, without specifying particular environmental criteria but requiring that afforestation should be for environmental purposes20. The net biodiversity benefits or otherwise of a particular afforestation scheme are primarily determined by the previous land use that is replaced by afforestation, the tree species planted, the age of the plantation, the management practices applied to the vegetation in between the young trees, the soil and other environmental conditions, and the distance from forest habitats that could be sources for colonisation of species. The net biodiversity benefit is likely to be influenced by whether the afforestation is being undertaken primarily for biodiversity, climate change mitigation or wood product production, as the objective will influence key management decisions such as which species to plant, the tree density and the rate of thinning. For example, plantations with a diverse mix of tree species are likely to provide more diversity of structure, habitat and food plants for fauna, whereas single species plantations of fast growing species are likely to be most effective for carbon sequestration or timber production but less valuable

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20 In 2007-2013 the measures excluding the planting of Christmas trees on any land; on agricultural land, support for afforestation of fast-growing species for short-term cultivation shall be granted for establishment costs only.
for biodiversity. Afforestation can also have an indirect impact on biodiversity through reducing the demand for timber and other forest products from existing forest (Brockerhoff et al, 2008), but indirect impacts were not considered in this review.

In some circumstances afforestation of agricultural land can be expected to have positive impacts on biodiversity, by restoring highly eroded land that has very low biodiversity, providing habitats for forest species in farmland poor in forest habitats, providing buffer zones between farmland and habitats such as water bodies, and/or connecting small isolated forest patches (Brockerhoff et al, 2008). It is however also possible that afforestation of semi-natural habitats with unique and/or rich biodiversity could result in a net loss of nature conservation value. Afforestation on intensively managed arable land or pasture is therefore more likely to provide a net biodiversity benefit than afforestation on extensively managed farmland. On extensively managed land, the gains in terms of forest species may not be as great as the losses of farmland and open land species, and that is often the type of land that is most likely to be afforested because of the lower economic opportunity costs.

The CAP funding regulations specify that care should be taken to avoid afforestation harmful to the biodiversity or causing other environmental damage. However, there is a lack of information on how much afforestation funded by RDP measures up to 2013 took place on semi-natural habitats with potentially high nature conservation value. In most Member States no data are available on the previous use of land afforested under the RDP measure, nor are there records of the presence of Annex I habitats or EU protected species on afforested land, or of landscape context, planting or maintenance (Elbersen et al, 2014). It is therefore not possible to draw conclusions on whether afforestation up to 2013 has actually had predominantly positive or negative impacts on EU protected habitats and species (Elbersen et al, 2014). Some Member States reported unquantified benefits for forest birds or Brown Bear, and some reported loss of non-forest Annex I habitats to afforestation (see Box 2-37).

**Box 2-37 Evidence from Member States of the importance of afforested areas to biodiversity**

A survey of experts in the Member States that have made significant use of the RDP afforestation measure found that those that targeted it to arable and other cropped land reported no negative effects (Denmark, Poland, Romania)\(^{21}\) (Elbersen et al, 2014). Three Member States reported unquantified benefits for forest birds from the creation of new native woodlands (Ireland, UK, Spain) and Spain reported unquantified extension of forest habitat in areas with Brown Bear populations. The same survey reported unquantified evidence that some non-forest Annex I habitat areas (as well as semi-natural grazed habitats more generally), were lost to EAFRD-funded afforestation since 1992 in the Czech Republic, Estonia, Hungary, Ireland, Lithuania, the UK and Spain, although the extent and impact of this is not established (Elbersen et al, 2014).

Plantations will gradually change from habitats dominated mostly by open land species to dominance of species tolerant of shade, depending on the speed of canopy closure, providing habitats increasingly suitable for forest species. Forest specialists will colonise new plantations at different rates dependent both on the species dispersal abilities and the distance of the plantation to existing forests with high species richness. Plantation forests can accelerate forest succession on previously deforested sites and abandoned agricultural areas where persistent ecological barriers to succession might otherwise preclude re-establishment of native species (Brockerhoff et al, 2008).

EFA plantations, which are less than 15 years old, will most likely be dominated by the open land vegetation that was present before afforestation and/or by generalist species that rapidly colonised the area after tree planting, depending on what management practices are used, as the trees are unlikely to be large enough to significantly influence the ground vegetation or micro-climate. This will depend, however, on the tree planting density used and the rate of growth of the selected species.

\(^{21}\) 15 Member States have made significant use of the measure since 1992, and only five have an area afforested under the CAP that is above 1% of the UAA (Portugal 8.41% of UAA afforested, Ireland 5.06% of UAA afforested, Spain 2.83% of UAA afforested, UK 1.73% of UAA afforested, Hungary 1.6% of UAA afforested).
Wild native plants

According to a global review, afforestation on degraded pasture land or arable land generally results in an increase in plant species richness, but afforestation on scrubland or semi-natural grassland generally results in a decrease in plant species richness compared to unforested habitats (Bremer and Farley, 2010), for example 5 years after afforestation of wet grassland in Ireland (Buscardo et al, 2008), 25 years after afforestation of Mediterranean garrigue scrub in Italy (Salvatore, La Mantia and Rühl, 2012), and 30-40 years after afforestation of Mediterranean semi-arid scrub in Spain (Gómez-Aparicio et al, 2009). However, afforestation on heath and scrubland are generally on land characterised by steep gradients and/or shallow soils not suitable for arable farming, and therefore are unlikely to be used for EFA.

Afforested land qualifying for EFA on previous arable or intensively managed grassland is likely to be dominated by generalist plant communities that have grown up from the seed bank or colonised from nearby farmland habitats, similarly to short rotation coppice (see evidence above). If the vegetation between the planted trees is managed with low intensity, this is likely to be dense, tall herbaceous vegetation with relatively low species diversity but some opportunities for the development of flowering perennials that might have a low abundance on intensively managed arable farmland. However, young (14-18 year old) plantations of native broadleaved trees on arable land close to older forests in Sweden were colonised relatively rapidly by some forest plants, including several nationally red-listed forest plant species (Brunet et al, 2012).

Box 2-38 Evidence of the impacts of afforestation on wild plant species diversity of semi-natural open habitats

A global meta-review of studies of afforestation that included paired comparisons between afforested and non-afforested habitats found that the value of plantations for plant species richness varies considerably depending on whether the original land cover is grassland, shrubland, primary forest, secondary forest, or degraded pasture, and whether native or exotic tree species are planted (Bremer and Farley, 2010). Plantations are most likely to contribute to plant species richness when established on degraded lands rather than replacing natural or semi-natural habitats, and when native tree species are used rather than non-native species.

Afforestation of Atlantic grassland

A study investigated the impact on plant diversity of afforestation of improved and wet grasslands with Sitka spruce (Picea sitchensis) in Ireland (Buscardo et al, 2008). Five years after tree planting, vascular plant species and Shannon’s biodiversity index were lower in the planted sites than on unplanted wet grassland sites. Bryophyte species richness was higher in afforested improved grassland sites than in unplanted sites. Competitive and vigorous grasses were more abundant, as were generalist species found in both open and wooded habitats, while small-stature shade-sensitive species were less abundant than on unplanted sites. The study concluded that the changes were primarily the result of the exclusion of grazing, ground preparation, changes in nutrient management and drainage for afforestation. Drainage reduced the diversity of species dependent on wet conditions, though drainage ditches provided temporary habitat for less competitive species.

Afforestation of Boreal heathlands

A series of case studies of afforestation with conifers on heathlands in Iceland and Denmark found an increase in the number of plant species during the first years after afforestation compared to open heathland, but as the canopy closed the heathland plant communities were replaced with shade tolerant plant groups with lower plant species richness (Halldorsson, Oddsdottir and Sigurdsson, 2008). The vascular plant species richness in coniferous plantations in Iceland and Norway was lower than in native broadleaved forests.

Afforestation of semi-arid Mediterranean habitats

A study investigated the impacts on plant diversity of afforestation of degraded species-poor garrigue vegetation and semi-arid grasslands with Pinus halepensis in Sicily, Italy (Salvatore, La Mantia and Rühl, 2012). After 25 years, plantations with the highest canopy cover had the lowest understory species richness and presence of species of biogeographical/conservation interest compared to the other plantations. The plantations with low density of canopy had flora with similar biogeographic values to unplanted garrigue. Compared to the garrigue sites, the plantations hindered the expansion of native woody species cover, notably of the garrigue climax species Quercus ilicis. The soil treatment associated with tree planting was probably the main factor influencing the dominance of an invasive alien species in the ground flora of plantations with a closed canopy (Oxalis pes-caprae). A study of afforestation with pines on semi-arid Mediterranean scrub in southern Spain (Gómez-Aparicio et al, 2009) found that 25 to 40 year old dense plantations had lower herbaceous plant species richness than unforested scrub, but pine plantations had higher recolonization by Quercus ilex because of targeted seed distribution by jays, and the moderately dense pine plantations at lower elevations promoted the recruitment of most plant species. A study of afforestation of natural grasslands in semi-arid environments in Jordan, in the Middle East, found that it resulted in reduced plant diversity and a marked change in species composition (Alrababah et al, 2007).
Practices associated with the CAP greening measures on climate

Box 2.39 Evidence of the impacts of afforestation on forest specialist wild plant species diversity

**Development of understory vegetation with age of stand**

A study looked at the development of the understory vegetation after afforestation of arable land with temperate broadleaved oak in Sweden (Brunet et al., 2011). Open-land species richness decreased as trees matured, while the richness of woody species and of generalist species remained stable. In plantations close to mature forests, a minimum of around 60 years was needed to accumulate forest plant species abundance to a level comparable with core forests, but isolated plantations did not increase in plant species richness after reaching 20 years of age, and forest specialists failed to colonise these isolated plantations. Plantations that were close to mature forest showed a gradual decrease in abundance of generalists, probably due to competition from colonizing forest specialists, which seemed to benefit from the development of a shrub layer. Soil pH in post-arable stands remained consistently higher than in continuously forested stands, which maintained differences in species composition between the stands. In young (14-18 year old) plantations of native broadleaved trees on arable land close to older forests, some forest plants colonised relatively rapidly (3-11.5 m per year), including several forest plant species listed in the Swedish red list, such as Adoxa moschatellina, Bromopsis benekenii, Primula elatior and Veronica montana (Brunet et al., 2012).

**Plant species richness in native stands versus non-native tree species plantations**

A study compared the species richness of lichens and bryophytes in non-native tree species plantations compared to native tree species stands in two regions of the UK (Quine and Humphrey, 2010). In Scotland, non-native Sitka Spruce (Picea sitchensis) plantations had higher bryophyte species richness but much lower lichen species richness than native Scots Pine (Pinus sylvestris) stands. In England, non-native Norway Spruce (Picea abies) stands had much lower lichen species richness and slightly lower bryophyte species richness than native oak (Quercus robur) stands. The study notes that the non-native plantations were planted on previously forested land or close to existing woodland and were younger than the native stands (10 to 80 years). Other studies have shown that mature and old non-native Sitka Spruce plantations in the UK have rare fungal species presence comparable to native woodlands (Humphrey et al., 2008). A study compared forest plant species richness, diversity and evenness in plantations of native Pinus pinaster and non-native Eucalyptus globulus with patches of natural forest of Quercus spp. in NW Portugal (Proença et al., 2010). Forest plant species richness, diversity and evenness were higher in oak forest than in pine and eucalypt plantations, and some forest species were exclusively observed in oak forest. Another study found similar diversity and composition of understory vegetation in young eucalypt plantations compared to shrubland, but the lowest diversity in intermediate aged plantations compared to shrubland and native forest patches, with plant diversity increasing in mature plantations (Calviño-Canela, Rubido-Bará and van Etten, 2012).

**Invertebrates**

Afforested areas will have increasingly different and more abundant soil invertebrate species composition as they age and the soil structure and litter changes compared to arable soils, and the litter dwelling surface invertebrate fauna will change in response to increased shading and the moister micro-climate under trees. Invertebrate predators such as carabid beetles and spiders are also likely to be different in species composition and abundance. However, unless the afforested area is directly adjacent to the arable area, it is unlikely to provide any ecosystem services or alter the biodiversity on the farmland as a whole, as the invertebrate species are likely to remain in the afforested area.

Box 2.40 Evidence of the impacts of afforestation on soil macro-invertebrates

A series of case studies of afforestation on heathlands in Iceland and Norway found that species richness of Collembola and earthworms increased under plantations as they aged compared to open heathland (Hallodsson, Oddsottir and Sigurdsson, 2008). The surface invertebrate fauna of spiders and beetles also remained similar to the heathland fauna during the first 10 years but then at the thicket stage was very different, due to the shady humid conditions. Non-native Sitka Spruce plantations in the UK have increasing abundance of specialist forest carabid beetles and decreasing generalists as they develop into mature stands (Mullen et al., 2008).

Afforested areas may provide suitable breeding and foraging habitat similar to field margins if they contain diverse herbaceous vegetation between the young trees. As the trees grow, butterflies are likely to become less diverse until forest-specific species are able to colonise. However, unless the afforested area is directly adjacent to the arable area, it is unlikely to alter the Lepidopteran biodiversity on the farmland as a whole, as the invertebrate species are likely to remain in the afforested area.

The edges and open spaces within afforested areas are likely to provide suitable nesting and hibernation habitat for pollinators and diverse flowering resources from flowers and shrubs. However,
if the afforestation is very dense and uniform in tree structure, it will lack these habitats and therefore provide little benefit to pollinators.

**Common farmland birds**

Forest plantations may increase overall bird density in farmland landscapes with a low proportion of woody habitats, but may also have a negative impact on birds specialised in open farmland landscapes (see Box for evidence). In other publications it is suggested that population declines of birds in arable landscapes of south and central-eastern Europe are linked to land abandonment and afforestation (Vorišek et al, 2010).

**Box 2-41 Evidence of impacts of afforestation on common farmland birds and other bird communities**

*Common farmland birds and afforestation*

A study assessed bird abundances at varying distances from the forest edge in mixed farming landscapes with variable amounts and spatial configurations of forest plantations in southern Portugal (Reino et al, 2009). The study concluded that forest plantations may increase overall bird density and abundance in adjacent farmland, at the expense of steppe birds of conservation concern. Overall bird density of woodland, farmland and ground-nesting birds declined with distance away from forest edges, and five of the nine most widespread and abundant farmland species were more abundant nearer to forest edges (Galerida larks, Stonechat, Linnet, Goldfinch and Corn Bunting). Afforestation on wet grassland and improved grassland in Ireland was associated with higher bird density and species richness (Wilson et al, 2012), and the authors concluded that afforestation on intensively managed grassland (grazing and/or cutting) is likely to benefit birds, especially in areas with relatively low levels of shrub cover.

*Birds in heathland versus afforested plantations*

A case study of afforestation on heathlands in Iceland found no major changes in the avifauna during the first 10 years of conversion from heathlands to non-native conifer plantations, although some forest birds appeared (redwing and redpoll), but by the time the stands were 20 years old (i.e. at the thicket stage) all the specialized open country birds had disappeared and the avifauna was composed of generalists (Haldorsson, Oddsdottir and Sigurðsson, 2008).

*Forest birds in native stands versus non-native plantations*

A study compared bird species richness, diversity and evenness in plantations of native Pinus pinaster and non-native Eucalyptus globulus with patches of natural forest of Quercus spp. in NW Portugal (Proença et al, 2010). Species such as Certhia brachydactyla, Sitta europaea and Dendrocopus major were common in oak and/or pine patches but were absent from eucalypt stands. Another study found the lowest bird species richness and abundance in Eucalypt plantations compared to shrublands, native forest and pine plantations, and this decreased with age of plantations due to the loss of shrubby vegetation providing flowers (Calviño- Cancela, 2013). The authors consider that this is likely to be due to the low presence of phytophagous insects on Eucalyptus globulus.

**Species that are the focus of the Birds and Habitats Directives**

**Box 2-42 Evidence of impacts of afforestation on species of conservation concern**

A study that assessed bird abundances at varying distances from the forest edge in southern Portugal found strong negative forest edge effects for two steppe bird species of conservation concern, Calandra Larks and Short-toed Larks, but not of Little Bustards and Tawny Pipits (Reino et al, 2009). The steppe birds tended to reach the highest species richness and abundances in large arable patches. Eucalypt plantations might also be associated with drastic population declines of Golden-striped salamander (Chioglossa lusitanica) in streams in Portugal22.

### 2.4.9 Catch crops / Green cover

Catch crops are generally grown over a short period from late-summer to the autumn after a main spring crop, with the intention of removing excess nutrients from the soil and reducing nitrate and phosphate leaching into water. Cover crops are generally planted after a late summer or autumn harvest, with the intention of reducing soil erosion and nitrate and phosphate pollution of water.

Grass undersown in the main crop may be used as a catch or cover crop. A positive biodiversity impact from catch and cover crops may arise in some sensitive areas from reduced soil erosion and nitrate leaching from the cropping surface into field margins and nearby water courses. Nitrate and phosphate leaching are a major driver of negative impacts on aquatic biodiversity (Grizzetti et al, 2011; Smith, 2003). However, the significance of these potential indirect effects on biodiversity in habitats beyond the arable field are highly uncertain and it was not possible within the scope of this review to attempt to quantify them.

The impact of catch and cover crops on biodiversity is strongly influenced by the way they are grown and managed, as well as by the species or species mix sown. Catch and cover crops may be cut and removed or incorporated into the soil. Soil incorporation can be carried out through tillage before the spring crop, but is now most often done through ‘burning down’ with herbicides (primarily glyphosate), especially where minimum or no tillage techniques are used. Catch and cover crops may replace bare soil or crop stubbles or they may replace a main crop in the crop rotation.

*Wild native plants*

Catch and cover crops are generally selected because of their capacity for vigorous and rapid growth of a vegetation canopy, thereby competing with and suppressing weed biomass and reducing weed flower and seed production. On EFA’s, catch and cover crops only qualify for the payment if they establish a sufficiently dense crop cover. They are therefore generally not expected to increase wild plant diversity or abundance in cropping systems. There is some evidence from a study that weeds in forage legume cover crops were less abundant than fallow in an organic system (see Box 2-43). However, this is not directly applicable to EFAs.

**Box 2-43 Evidence of the impacts of catch/cover crops on wild plant diversity**

*Legume cover crops*

An experiment in Greece compared Vetch (*Vicia sativa*) and Red Clover (*Trifolium pratense*) winter cover crops with winter fallow following wheat under organic management (Bilalis, Karkanis and Efthimiadou, 2009). The legumes contained lower weed density and biomass than the fallow, with the vetch significantly lower than Red Clover. The cover crops were not fertilised. The vetch developed a rapid vegetation cover, inhibiting the growth of most autumn-germinating weeds, whilst the Red Clover was slow to establish and was therefore a poorer weed competitor.

**Invertebrates**

Catch and cover crops that are incorporated into the soil are expected to contribute to increased soil organic matter, which is a driver of soil invertebrate macrofauna abundance. Legume cover crops provide high quality plant residues and root biomass for decomposers such as surface-feeding earthworms (Eisenhauer et al, 2009). Even if catch crops are harvested and removed from the field, they may have a beneficial effect on soil macrofauna by reducing soil erosion and retaining soil moisture over the summer period.

Cover crops are unlikely to provide abundant alternative prey or hosts for natural enemies of crop pests as they are only present in the field for a short period or only overwinter, and do not flower. However, undersown grass or winter cover crops can provide hibernation habitat and increase larval survival of generalist ground predators, providing a more hospitable environment than bare soil. If summer catch crops are allowed to flower they can provide prey or host and/or nectar or pollen resources that may carry over larger predator populations into the next crop or help to maintain populations throughout the season, compared to bare fallow23. A literature review (Holland et al, 2016) found evidence from three studies that undersown grass can enhance in-field carabid beetle and spider abundance in arable fields in Europe, but found no evidence for spill-over of natural enemies into crops from undersowing and cover crops, and no evidence of improved pest control. A

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key determinant of the impact of cover crops on carabid beetle abundance in crops is tillage frequency and timing, as well as the crop micro-climate (Holland and Luff, 2000).

Adult **butterflies** of generalist mobile species will visit some summer flowering catch crops for nectar, if they are allowed to flower profusely and if they contain a mix of plant species, which might increase the egg laying rate of some species (Settele et al, 2009). Some common cover crops in the Brassica family (mustards, charlock, radish etc.) are larval food crops for pierid butterflies (Pieridae), but these are the most common and ubiquitous species and generally regarded as crop pests, and so are not considered to be of conservation importance. Because catch and cover crops generally reduce weed abundance and diversity, they may have an overall negative effect on other common Lepidoptera on arable farmland that use weeds as larval plants or nectar sources. However, no direct evidence was found to substantiate this.

Winter cover crops are not expected to have any direct impact on **pollinators** as bees and hoverflies rely on semi-natural habitats for the winter period24. There is some evidence from the UK that overwinter stubbles provide early-season resources from flowering weeds for bumblebees and sawflies (Symphyta), especially when followed by spring fallow (Evans, Armstrong-Brown and Grice, 2002), and replacing them with winter cover crops could worsen the situation for pollinators if flowering weeds are less available.

The benefits of summer catch crops to pollinators will depend greatly on the length of time the crop flowers and the type of flower (see nitrogen-fixing crops for the evidence on green manure legumes). A study in Germany found that summer clover-grass leys had only half as many nesting solitary bees as fallow with naturally regenerated vegetation (Gathmann, Greiler and Tscharntke, 1994). Brassica species such as mustard or radish are attractive to a wide range of flower visiting bees, hoverflies and non-syrphid flies (Wood, Holland and Goulson, 2016a), and certain flowers such as Phacelia are very nectar-rich and attract high abundances of flower visiting insects. However, there is no evidence that this has an impact on the population level as it may simply temporarily draw foraging individuals away from other habitats (see nitrogen-fixing crops section for more discussion).

**Common farmland birds**

Catch and cover crops are unlikely to provide good foraging habitats for seed-eating farmland birds given their typically low weed densities, and they are normally cut before setting seed. They are probably therefore generally detrimental if they replace stubbles which are richer in seeds, as for example found in a study in Poland (Golawski et al, 2013). But this impact is difficult to quantify as stubbles of conventionally managed arable crops now provide limited seed reservoirs due to the reduction of weeds through tillage and herbicide use, and stubbles are frequently ploughed soon after harvest. Significant benefits for seed-eating birds could therefore arise if winter cover crops comprise seed-bearing species that are favoured by farmland birds and are allowed to set seed, but this is unlikely to be common practice on EFAs.

As catch and cover crops are likely to increase soil invertebrates, this may provide increased food resources for some birds especially immediately after the crop is ploughed in. Some catch crops may provide suitable breeding habitat for some birds and those that hold high densities of invertebrates (e.g. mustard, radish, Phacelia) may provide good foraging habitats. However, no evidence was found on any of these potential biodiversity benefits.

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**Box 2-44 Evidence of impacts of cover/catch crops on farmland birds compared to winter stubbles and fallow**

A Europe-wide study at the landscape scale (Geiger et al, 2010) observed more farmland birds foraging in winter in landscapes with higher proportions of green manure cover crops, overwinter stubble, and pasture, but did not differentiate between these crop types. A study in low-intensity arable farmland in Poland (Golawski et al, 2013) compared winter foraging birds on winter catch crops of mustard, on cereal stubbles and on bare soil. The study recorded 28 species, of which the most numerous were Yellowhammer (*Emberiza citronella*), Corn Bunting (*Emberiza calandra*), Tree Sparrow (*Passer montanus*) and Shorelark (*Eremophila alpestris*). The stubbles supported a relatively greater density of birds than the mustard winter catch crop and

24 Crops must remain until March in Romania, but due to the continental climate it is unlikely that bees will emerge from hibernation or legumes such as clover flower before the end of March.
ploughed fields. The study concluded that increasing the area of winter catch crops instead of leaving fields bare and ploughed may favour birds, but when winter crops are sown in fields that would otherwise be left as stubble, wintering conditions for birds deteriorate, especially for buntings (see review of fallow in the next section). A study of winter foraging birds in England (Stoate, Szczur and Aebischer, 2003) found that birds were most abundant on wild bird cover crops compared to other crops on the farm (but did not compare overwinter cereal stubbles or fallow). The bird cover sown with kale (Brassica napus) and quinoa (Chenopodium quinoa) was used by Redpoll, Tree Sparrow, Reed Bunting, Song Thrush, and Linnet, whilst Yellowhammer and Goldfinch mainly used the cereals and linseed mix. Farmers in the Netherlands are being encouraged to plant Black Oat (Avena strigosa) as a cover crop because of its value as winter feeding habitat for granivorous birds. However, flax (Linum usitatissimum), which is being promoted as a catch crop in some regions, was not preferred by Skylarks and Yellowhammers for establishing territories relative to autumn-sown crops on lowland farmland in Northern France (Tolhurst et al., 2014).

Species that are the focus of the Birds and Habitats Directives

Little evidence of the possible impacts on species that are the focus of the Birds and Habitats Directives was found (see Box 2-45). However, it is likely that, as for common farmland birds, losses of cereal stubbles to autumn sown cover crops would reduce food resources for some species, such as the Little Bustard (Faria and Silva, 2010). Such losses of stubbles could also have negative impacts on some plant weeds. On the other hand cover crops could provide additional plant foliage food resources in the autumn compared to bare soil.

<table>
<thead>
<tr>
<th>Box 2-45 Evidence of impacts of catch/cover crops on species that are the focus of the Birds and Habitats Directives</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Little Bustard</strong></td>
</tr>
<tr>
<td>A study in southern Portugal found that Little Bustard preferred cereal stubbles and vegetated fallow lands over crops and ploughed lands during the autumn, which were characterized by the growth of fresh green weeds (Faria and Silva, 2010). However, the study also found that fallow has decreased in area and most cereal stubbles were being ploughed in rather than left.</td>
</tr>
<tr>
<td><strong>European Hamster</strong></td>
</tr>
<tr>
<td>A project in France is testing the use of spring legume mixes undersown in winter wheat on the European Hamster population (LIFE Alister project, 2016), but no results are available yet.</td>
</tr>
<tr>
<td><strong>Rare arable weeds</strong></td>
</tr>
<tr>
<td>Ploughing of cereal stubbles directly after harvest is a threat to the rare EU protected arable weed Notothylas orbicularis (IUCN, 2015).</td>
</tr>
</tbody>
</table>

2.4.10 Nitrogen-fixing crops

Nitrogen-fixing crops (i.e. plants in the Fabaceae family, referred to in this report as legumes) are grown as a single crop or mixed legume crop during the spring and summer cropping period, according to the EFA specifications defined by the regulations and by Member State rules. This review does not address the impact of nitrogen-fixing shrubs or trees as these are only currently eligible on EFAs as part of some agroforestry systems.

The impact of nitrogen-fixing crops on biodiversity is influenced by the way they are grown and managed. They can be divided into crops grown for their grain (e.g. field peas, faba beans, lentils and soyabeans) and crops grown as green forage and/or green manure (e.g. Lucerne/alfalfa, sainfoin, vetches, clovers). Cultivation practices, including cropping period, fertiliser and pesticide use, cutting and tillage are different for these two groups. A positive biodiversity impact may arise from reduced fertiliser use in crop rotations with nitrogen-fixing crops in some sensitive areas due to reduced nitrate leaching from the cropping surface into field margins and nearby water courses. Nitrate and phosphate leaching are a major driver of negative impacts on aquatic biodiversity (Grizzetti et al., 2011; Smith, 2003). However, nitrogen-fixing crops can result in increased nitrate leaching if the crop harvest is followed by soil erosion. The significance of these potential indirect effects on biodiversity in habitats beyond the arable field are highly uncertain and it was not possible within the scope of this review to attempt to quantify them.

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Wild native plants

Legumes, being broad-leaved (dicotyledonous) plants, compete with non-crop plants for space, water and nutrients in ways that contrast with grass-type cereal crops, so their weed populations can be expected to differ from cereals. However, weed densities and species composition within crops are generally more heavily influenced by tillage, herbicide use and soil type than by crop type or diversity of crops in the rotation (Ulber et al, 2009).

The review found little evidence that weed diversity in conventionally managed legume crops differs from other conventionally managed crops (Box 2-46). One study showed a shift towards broad-leaved perennial weeds in multi-annual forage alfalfa crops (Meiss et al, 2010a), and these weed species tend to provide more food resources for flower-visiting insects. Another study found more diverse weeds in alfalfa managed without herbicides or summer cutting (Badenhausser et al, 2008). Only one study differentiated weed groups with benefits for wildlife, and it found a higher coverage of flowering plants that provide bumblebee foraging in the margins and crop edges with organic or low input legume-containing rotations (clover and other legumes) (Marja et al, 2014). However, these cases are not typical for most EFAs as they involve reduced or no herbicide use to control weeds before sowing and no summer cutting.

Grain legume crops are liable to high weed densities due to their long germination period and slow initial growth compared to cereal crops, but as their cultivation involves soil tillage and/or pre-sowing herbicide use, this is expected to keep the weed density low. Studies in Finland and Germany found that under conventional management, the weed flora of spring cropped field pea did not differ significantly from other spring crops on similar soils (Andreasen and Skovgaard, 2009; Gathmann, Greiler and Tscharntke, 1994). It can be expected that this will also apply to grain legume crops grown in conventionally managed intensive arable rotations across Europe, in which weed densities are generally kept very low, as shown by a survey of wild plant coverage in arable fields in 39 regions in 10 European countries which found wild plant coverage below 1% except in two regions (IFAB, 2015). If faba bean is harvested late in the season, this can allow perennial weeds to flower (Köpke and Nemecek, 2010), providing resources for flower-visiting insects. No evidence was found regarding the weed flora in soybean crops.

Under organic management, the weed flora of field pea is more species rich and has a higher biomass than in field pea or spring cereals pre-treated with herbicide (Deveikyte, Kadziuliene and Sarunaite, 2009; Graziani et al, 2012; Salonen, Hyvönen and Jalli, 2005). Minimum tillage had little influence on weed density in a legume-containing crop rotation managed intensively with herbicides and fertiliser over two decades (Hernández Plaza et al, 2011), whilst over a shorter period it reduced diversity in a low-input legume-containing rotation (Santín-Montanyá et al, 2014).

<table>
<thead>
<tr>
<th>Box 2-46 Evidence of weed flora abundance and diversity in nitrogen-fixing crops</th>
</tr>
</thead>
<tbody>
<tr>
<td>Field pea (Pisum sativum)</td>
</tr>
<tr>
<td>A survey of the weed flora of 11 crop types across Denmark (Andreasen and Skovgaard, 2009) under conventional intensive management found that the weed flora of spring cropped field peas was very similar to spring barley and spring rape but differed significantly from winter cereal crops, and from spring grown beet and maize. In a German study, field pea crops contained a slightly higher mean plant species richness than rye fields and slightly lower than barley fields after one growing season (Gathmann, Greiler and Tscharntke, 1994). The study did not record how the pea crops were managed.</td>
</tr>
<tr>
<td>A study that compared weeds in field pea fields in southwestern Finland under conventional management with herbicide and organic fields that mostly used no pre-crop weed management following spring cereal (Salonen, Hyvönen and Jalli, 2005) found 59 weed species under the conventional management, with an average of 10 species per field, and 68 under the organic management, with an average of 18 species per field. An experiment in Lithuania found that weed biomass in field pea managed without the use of herbicides or inorganic fertiliser was significantly higher than in the spring grown cereal crops and total weed number was 1.3 to 1.6 fold higher (Deveikyte, Kadziuliene and Sarunaite, 2009). In a crop rotation with field pea in Mediterranean Italy (Graziani et al, 2012), the adoption of organic management resulted in a significant increase in weed seedbank density, particularly four competitive summer weed species (Portulaca oleracea L., Amaranthus retroflexus L. and Chenopodium album L.), compared to a low-input but non-organic management.</td>
</tr>
<tr>
<td>An experiment of a low input dry pea crop-winter wheat-barley-fallow rotation in semi-arid Spain found that over four growing seasons weed density was highest in minimum tillage than in conventional and zero-till systems, in which weed density decreased (Santín-Montanyá et al, 2014). It is likely that minimum tillage facilitates the germination of weed seeds in the upper soil layer. Pea plots under zero-tillage had less diverse and less balanced weed communities compared to the other forms of tillage, mainly due to the dominance of Ryegrass (Lolium rigidum) which was favoured by the higher soil humidity. This annual grass matures at the same time as the pea and its seeds are removed and redistributed on the field through the harvesting process.</td>
</tr>
</tbody>
</table>

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Invertebrates

Legumes can be attractive to soil macro-invertebrates because of the input of nitrogen rich residues into the soil and because legume roots stimulate soil microbial activity. There is evidence that soil macro-invertebrates are more abundant under forage/green manure legume crops than non-legume crops (Box 2-47). Multi-year forage legume crops are characterized by reduced belowground disturbance due to the absence of frequent tillage (which is strongly detrimental to earthworms) (Krogh et al, 2007; Nieminen et al, 2011), and the accumulation of plant litter on the soil surface, which favours detritivores. For example, alfalfa leys had higher mean population densities and biomass of above ground arthropods, particularly detritivores/microvores, than barley and grass ley crops (Curry, 1986). There were more surface-feeding (anecic) earthworms under four-year galega than under a grass crop and bare fallow (Epie, Cass and Stoddard, 2015), and more earthworms under four-year clover than under grass (Crotty et al, 2015).

Insecticides and fungicides applied to grain legume crops can negatively affect soil macro-invertebrates. Earthworms are highly susceptible to neonicotinoid insecticides that are currently widely used in arable crop rotations in the EU (Chagnon et al, 2015), and residues are likely to be present in the soil even though their use is currently banned on legume crops. Earthworms are also affected by other insecticides and fungicides through both sublethal and lethal effects (Jänsch et al, 2006; Tu et al, 2011).

Box 2-47 Evidence of impacts of nitrogen-fixing crops on soil invertebrates

Alfalfa (Medicago sativa)

Alfalfa leys in an experiment in Sweden had higher mean population densities and biomass of above ground arthropods than barley and grass ley plots, with a higher share of the biomass made up by detritivores/microvores compared to the herbivore and predator biomass (Curry, 1986).

Galega (Galega orientalis)
An experiment in Finland (Epie, Cass and Stoddard, 2015) found more earthworm numbers and biomass (Aporrectodea caliginosa, A. rosea, Lumbricus rubellus, and L. terrestris) under plots of pure four year stands of the legume crop galega (Galega orientalis), compared to a mixture of galega and reed canary grass, pure stands of Reed Canary Grass (Phalaris arundinacea) and bare fallow, although the increase was statistically significant only in summer.

Clovers (Trifolium spp)
A UK experiment (Crotty et al, 2015) found higher earthworm abundance and biomass under four year plots of White Clover (Trifolium repens) than under plots of Ryegrass (Lolium perenne) or Chicory (Chicorium intybus), especially the anecic species. Nematode fungal feeders and detritivorous Collembola (Poduromorpha) dominated under the Red Clover (Trifolium pratense) and White Clover and Chicory, whilst nematode herbivores and herbivorous Collembola (Symphypleona) dominated under the Ryegrass.

Some legumes have a high structural complexity, with climbing and creeping growth forms, compared to cereal crops, which may make them more attractive to canopy-dwelling invertebrate predators such as spiders. Faba bean and vetches (Vicia species) offer extra-floral nectaries on the underside of the stipules, which are accessed by invertebrate natural enemies of pests, such as parasitoid wasps, ants, and bugs. There appears to be little evidence of actual impacts, but two studies showed that uncut alfalfa provides good habitat for weed seed predating Carabid beetles (Meiss et al, 2010a) and for spiders (Samu, 2003), but did not show whether this increased their abundance in the main crop or whether it influenced pest predation (Box 2-48). Green manure and forage legumes are more likely to contain invertebrate predators such as carabid beetles, as a key determinant of carabid beetle abundance in crops is tillage frequency and timing, as well as the crop micro-climate (Holland and Luff, 2000).

If nitrogen-fixing crops are not treated with insecticide, they have the potential to build-up invertebrate predator and parasitoid densities and carry over natural enemy populations into the subsequent arable crop, providing enhanced natural biological control. However, the potential benefits of grain legumes for natural enemies of crop pests is likely to be limited by conventional farming practices including pesticide and fertiliser use (Kovács-Hostyánszki, Batáry and Báldi, 2011). A comparative study in England (UK) (Holland et al, 2012b) found a lower total biomass of invertebrates in commercial faba bean and pea crops than in cereal crops, indicating that despite the variety of insects that feed on legumes, a high level of insecticide use is suppressing invertebrate abundance.

**Box 2-48 Evidence of impacts of nitrogen-fixing crops on invertebrate predators of pests**

**Alfalfa (Medicago sativa)** Uncut alfalfa provides good foraging habitat for seed-eating Carabid beetles compared to cut alfalfa and grass and bare soil (Meiss et al, 2010a), as shown by a French experiment using seeds of three common annual weed species (Alopecurus myosuroides, Sinapsis arvensis and Viola arvensis). An experiment (Samu, 2003) in Hungary found that uncut alfalfa plots had an average of over one and a half more spiders than continuously-cut alfalfa plots over three years. It is likely that spiders are attracted to the unmown strips from the mown strips and also from other habitats. However, the presence of unmown strips did not increase spider abundance in the cut strips in between. Five spider species made up over three quarters of all spider numbers in both cut and uncut Alfalfa, but spider species diversity was greater in the unmown strips.

Little direct evidence was found of the use of nitrogen-fixing crops by butterflies and moths. As indicated in Box 2-49, some species use legumes as their larval host plants, and can reproduce if alfalfa or clover is managed extensively with delayed mowing and low inputs (Loos et al, 2014; Manil and Chague, 2014). Adult butterflies collect nectar on flowering legume crops when they are available, and for some species, adult foraging can have an impact on population size by increasing female egg productivity and male fecundity (Settele et al, 2009). However, butterflies are negatively affected by neonicotinoid insecticide use (Gilburn et al, 2015) and may be affected by insecticide residues in the legume flowers and weeds originating from previous treated crops. Although neonicotinoids are currently banned on legume crops, they are still widely used as seed treatments on winter cereals which may precede the EFA crop. Experimental evidence has shown that traces of neonicotinoid insecticides and various fungicides are present in the pollen of oilseed rape and of weeds growing in the field margins of treated crop fields (Botías et al, 2015; David et al, 2016). Such levels of these insecticides have been shown to have detrimental impacts on some bees (see below), and may also affect butterflies (Gilburn et al, 2015).
Nitrogen-fixing crops on EFAs increase the presence of mass-flowering crops on arable farmland at the landscape scale, which can influence the highly mobile flower visiting insects including bumblebees (Box 2-50). The commoner bumblebee species are influenced by landscape wide effects because they will range up to 5 km around the nest if there is insufficient food available nearby (Greenleaf et al, 2007; Osborne et al, 2008). Honeybees are also highly attracted to mass-flowering crops (Montero-Castaño, Ortiz-Sánchez and Vilà, 2016; Rollin et al, 2013), but are not further considered in this literature review as they are not wild species.

Solitary bees as a group are less likely to respond to mass-flowering crops than bumblebees, as they have shorter foraging ranges and tend to have more specialist flower preferences for various plant groups (Rollin et al, 2013), whilst legume species are the major pollen source for bumblebee species (Goulson et al, 2005). The majority of solitary bees forage within a few hundred metres of their nest site, although some individuals of the larger species are capable of foraging up to 1.5 km from the nest (Zurbuchen et al, 2010). This makes them more dependent on local conditions and less sensitive to landscape-scale crop composition (Gathmann and Tscharntke, 2002; Holzschuh et al, 2011). Solitary bees are dependent on the presence of undisturbed patches of semi-natural vegetation in which to nest and on the constant presence of flowers offering nectar and pollen within their foraging range, and are therefore unlikely to benefit significantly from nitrogen-fixing crops on EFAs (except multi-year alfalfa). For example, one study in Germany found no effect of oilseed rape crops on solitary bee richness and abundance in the field margins (Kovács-Hostyánszki et al, 2013).

Field peas have very limited nectar production (0.96 kg/ha) compared with cultivated beans (52 kg/ha), Birdfoot Trefoil (50 kg/ha) and alfalfa (126 kg/ha), with all comparing poorly with red clover at 894 kg/ha (Baude et al, 2016). Some soybean cultivars have been bred to self-pollinate, with flowers offering little nectar and a large proportion of flowers that do not open for flower visiting insects at all, but cultivated soybean cultivars have significant differences in flower attraction and reward traits and some are heavily visited by bees (Milfont et al, 2013; Suso et al, 2016). Beans and peas have deep flowers that are only effectively accessed by bumblebees and some Lepidoptera. These crops therefore act as a strong filter on flower-visited insects by providing bumblebees with a food resource that is relatively free from flower visiting competitors. However, bumblebees and the larger solitary bees often cut holes in the flower base, providing access for short-tongued bees and other nectar seeking insects, so faba bean is visited by a more diverse range of flower-visiting insects if these large bees are already present (Tasei, 1976, cited by (Garratt et al, 2014).

A number of studies indicate the attractiveness of faba beans to foraging bumblebees during the 3-4 weeks in which they are flowering on EFAs, but there is little evidence that the crops are contributing to population growth. After the crop harvest, the survival of bumblebee colonies depends on the availability of weeds in and around crops, flowering plants in field margins and hedges, and the presence of semi-natural grassland or scrub and woodland patches. Consequently, bumblebees have been observed to decline sharply in simplified landscapes after late July if the availability of herbaceous flowers is low (Persson and Smith, 2013). Mass-flowering crops provide a significant boost to bumblebee nests that have survived up until they flower in May and June, but in an intensive arable landscape with few non-cropped areas many colonies will not survive until then. Mass-flowering crops may therefore be temporarily attracting large numbers of workers from a small number of nests (Herrmann et al, 2007), which does not result in population growth in the next year. Even when colonies produce more bumblebee workers as a result of the crops, this does not necessarily result in an increase in the number of queens at the end of the season, which is a determining factor for colony number in the following year (Westphal, Steffan-Dewenter and Tscharntke, 2009).
Forage legume species show a greater variability in flowering time than grain legume crops. They may therefore have a more lasting beneficial effect on wild bee populations than grain legumes, particularly if grown for a whole year or longer and allowed to flower regularly (Kovács-Hostyánszki et al, 2016), although this is unlikely to be the case on most EFAs. There is some evidence that late season red clover fields have a population level effect on bumblebees (Rundlöf et al, 2014), by increasing queen production. Forage legumes may also promote small mammal populations, which provide nesting sites in their burrows for ground-nesting bumblebees (e.g. *Bombus terrestris*).

Nitrogen-fixing crops in conventionally managed arable crop rotations can contain traces of systemic pesticides, such as neonicotinoid insecticides, that were applied to the preceding winter crop and that the plants have taken up from residues in the soil. Neonicotinoids have been shown to have sublethal effects on bumblebees at concentrations similar to those found in the study (Whitehorn et al, 2012), and also have sublethal effects on solitary bees (Sandrock et al, 2014).

**Box 2**

**50 Evidence of impacts of nitrogen-fixing crops on pollinators**

Faba bean (*Vicia faba*)

A study of faba bean in the UK (Garratt et al, 2014) found the vast majority of flower visits (including raids) were by six species/species groups of bumblebee, representing the most common species. The flowers were also visited by a large number of non-syrphid flies, as well as hoverflies (syrphids), butterflies, beetles and sawflies. Another UK study of bumblebees foraging along field margins of faba bean crops (Hanley et al, 2011) found that although no single bumblebee species showed any consistent change in relative frequency in response to the presence of the crop, when taken across all years and bumblebee species combined, twice as many bumblebees were visiting flowers adjacent to the faba bean compared to other arable margins. However, two weeks after bean flowering, the bumblebee activity along hedgerows was no different between crops. There therefore seemed to be no long lived effect on bumblebee populations.

A study in Sweden (Andersson et al, 2014) found a higher pollination rate of faba bean on organic farms compared to conventional farms, and that pollination on organic farms increased the more semi-natural habitat there was in the landscape, indicating that bumblebee abundance was higher where there was more of these habitats. In contrast, on conventionally managed farms with fertiliser and pesticide use, the pollination was unaffected by either semi-natural habitat in the landscape or leys on the farm, indicating that bumblebee abundance on faba bean on these farms was constrained by other factors. A study of insects visiting faba bean in the UK (Carr et al, 2009) found that the abundance and diversity of wild bees in the crop (which were almost all bumblebees) increased in landscapes with a greater abundance of transitional woodland-shrub habitats. The area of these patches was correlated with wild bee abundance and their perimeter:area ratio was correlated with sub-generic diversity. Another UK study of bumblebees visiting faba bean (Nayak et al, 2015) found that foraging bumblebees were more abundant on the crop the more semi-natural habitat cover and flower abundance there was within the surrounding 2 km.

A UK study observed an increase in colony density of the long-tongued bumblebee *Bombus pascuorum* in the locality of faba bean crops within a 1km radius, as well as a significant effect of oilseed rape fields and non-cropped semi-natural habitat areas, in late July (Knight et al, 2009). Another UK study (Goulson et al, 2010) found very little correlation between mass flowering crops (oilseed rape and faba beans) and the nest density and nest survivorship of two bumblebee species (*Bombus lapidarius* and *Bombus pascuorum*), but did find a significant positive correlation with the area of urban gardens and grassland. Interestingly, one site (which was excluded from the statistical analysis) appeared to have approximately four times as many *B. lapidarius* nests in both early and late season samples, and approximately five times as many *B. pascuorum* in late samples, compared to the other sites, almost certainly because of a ~5-ha clover ley adjacent to this site.

Sulla (*Hedysarum coronarium*)

A study on the Mediterranean legume crop sulla (*Hedysarum coronarium*) on Menorca, Spain, found 9 wild bee species visiting sulla compared to 19 species on wild flowers in adjacent semi-natural scrubland patches (Montero-Castaño, Ortiz-Sánchez and Vilà, 2016). The single bumblebee species, *Bombus terrestris*, and one of the solitary bee species *Eucera numida*, were found only on the sulla and not in the scrubland, whilst seven species were present on both. Wild bee abundance was not significantly different in the crop versus the scrubland.

Alfalfa (*Medicago sativa*)

A study in traditionally managed diverse farming landscapes in Transylvania, Romania (Kovács-Hostyánszki et al, 2016), found that on arable land alfalfa provided a high density of flowers, which were well utilised by bumblebees (especially *Bombus terrestris*) and some oligolectic wild bee species such as *Andrena labialis*, *Eucera nigrescens*, *Melitta leporine*. Hoverflies showed high abundance in alfalfa and fallow. The alfalfa is managed for forage with very low inputs. Very little of the arable farmland in this region is subject to the EFA requirements due to the small farm size.

Red Clover (*Trifolium pratense*)

A Swedish study (Rundlöf et al, 2014) found higher bumblebee queen densities of six *Bombus* species in established late-season red clover fields than in linear field borders in the surrounding landscapes.
Common farmland birds
Nitrogen-fixing crops contribute to increasing crop diversity at the landscape scale. It might therefore be expected that this benefits common farmland birds, as increases in crop diversity can increase breeding and feeding opportunities for some species (Benton, Vickery and Wilson, 2003), such as observed for nesting Skylarks (Chamberlain, Vickery and Gough, 2000; Miguel, Gaucherel and Bretagnolle, 2013; Wilson et al, 1997). However, the structure of each crop and timing of their establishment is also important, so increasing crop diversity is not necessarily beneficial, as found by Chamberlain et al (Chamberlain et al, 1999) for Skylarks.

There appears to be no direct evidence in the literature of possible crop diversity related benefits and little evidence of other possible impacts of nitrogen-fixing crops on common farmland birds (Box 2-51). One study found that Skylarks achieve high reproductive success in a multi-year alfalfa crop if it is not mown more than twice a year, thereby retaining a suitable height for breeding throughout the year (Kuiper et al, 2015). However, alfalfa on EFAs may be cut more frequently than this, so breeding success rates would be expected to be low in such circumstances. Reproductive success of ground nesting birds is also expected to be low for conventionally managed grain legumes, as observed for Yellow Wagtail (Gilroy et al, 2011). Furthermore, one study found that invertebrate food resources required by chicks are less abundant on conventionally managed grain legume crops than on cereal crops (Holland et al, 2012b).

Box 2-51 Evidence of impacts of nitrogen-fixing crops on common farmland birds
Faba bean (Vicia faba)
A study in southern Poland found no particular effect on the farmland bird community of the loss of clover and faba bean crops from farming systems (Kopij, 2008). A UK study found that Yellow Wagtail (Motacilla flava) nested in faba bean crops but had a high nest failure rate from predation, compared to nests in cereal fields (Gilroy et al, 2011; Kirby et al, 2012). A UK study compared the availability of bird chick food invertebrates in different conventionally managed crops and found that numbers were lower on peas and beans than in the winter and spring cereals, and estimated biomass was also lower (Holland et al, 2012b).

Alfalfa (Medicago sativa)
A study of Skylark breeding success in crops in the Netherlands (Kuiper et al, 2015) found that the highest reproductive success was achieved in alfalfa compared to grassland and cereal crops. Nestling weight was significantly higher in alfalfa and grassland compared to cereals, but survival was low in grassland due to frequent silage cutting, whereas the alfalfa was mown twice a year and retained a suitable height for breeding throughout the year.

Species that are the focus of the Birds and Habitats Directives
The evidence regarding nitrogen-fixing crops and species that are the focus of the Birds or Habitats Directives refers only to alfalfa crops grown as green manure and fodder (Box 2-52). Multi-year alfalfa crops are good quality habitats for small mammals, such as Common Voles (Microtus arvalis), owing to their long-term stability and suitability for vole colony formation (as they remain unploughed for 5–6 years) and hold high densities even if regularly cut (Jareño et al, 2015; Rodríguez-Pastor et al, 2016). These are key prey items for birds of prey on the Birds Directive Annex II such as Montagu’s Harrier, Hen Harrier and Black-winged Kite.

Alfalfa also provides important habitats for the Little Bustard (Badenhauser et al, 2008; Bretagnolle et al, 2011a; Bretagnolle et al, 2011b; Lapièdra et al, 2011; Silva, Faria and Catry, 2007) and Great Bustard (Martín et al, 2012). Alfalfa crops are also an important feeding habitat for European Hamster (Cricetus cricetus) in intensively managed arable farmland (Albert, Reiners and Encarnação, 2011; O’Brien, 2015), but it is important to note that low intensity management is necessary to realise these benefits (with infrequent cutting, long cropping periods and little other disturbance). As mentioned above for other farmland birds, if these crops are cut during the breeding season then they can become ecological traps, leading to nest destruction and very high levels of egg and chick mortality.
Box 2.52 Evidence of impacts of alfalfa on species in the Birds and Habitats Directives

Alfalfa

Ortolan Bunting (Emberiza hortulana)
The occurrence of the Ortolan Bunting in Italy is linked to the presence of alfalfa crops and bare soil (Morelli, 2012).

Little Bustard (Tetrax tetrax)
A study of Little Bustard in southern France found that the provision of five year alfalfa crops in cereal rotations with no mowing between mid-May and end July has led to a sharp increase in female productivity (Bretagnolle et al, 2011b). Differences in grasshopper availability appear to be critical to productivity, as estimated by the number of fledglings counted in post-nuptial groups, but alfalfa crops had rather low grasshopper densities (about 0.8 individuals per m²) compared to other temporary grasslands (about 3.1 individuals per m²) (Bretagnolle et al, 2011a). Alfalfa crops that were not cut or treated with herbicide or insecticide between May and August had higher densities of immatures (0.07 to 17.4 m⁻²) and adults (0 to 5.66 m⁻²) than alfalfa treated with pesticides and cut in July (Badenhausser et al, 2008). A study of female Little Bustards on the cereal pseudo-steppe of the Spanish Lleida plains (Lapietà et al, 2011) found that unharvested alfalfa crops were preferred foraging habitats for sedentary females after the main cereal harvest, but wandering females predominantly shifted to field margins and bare fallows. However, females were mainly breeding in cereal fields, and the Lleida Plain population’s breeding success is similar to the lowest estimations for the endangered population in France and is not enough to guarantee the long term viability of the population.

Great Bustard (Otis tarda)

Alfalfa crops are very important foraging and nesting crops for Great Bustard (Magaña et al, 2010; Nagy, 2010), such that it has been estimated that in Spain it should represent at least 8% of the breeding area in order to provide ideal habitat for the species (Alonso and Alonso, 1990). Declines in the species are considered to be due in part to reductions in the area of alfalfa and other pulses (Rosell and Viladomiu, 2005). Increases in Great Bustard numbers in Castilla y León province in central Spain were found to be positively correlated with increases in the extent of unirrigated alfalfa and vetch crops grown with no cutting, fertilizer or pesticide use between May and August (Martin et al, 2012).

Birds of prey such as Montagu’s Harrier (Circus pygargus), Hen Harrier (Circus cyaneus) and Black-winged Kite (Elanus caeruleus)

In Spain (Rodríguez-Pastor et al, 2016), irrigated (and some unirrigated) multiannual (at least 5 year) Alfalfa crops contained higher densities of small mammals than cereal crops. Though alfalfa crops are cut at least four times during the summer, they provide a stable habitat for burrowing colonies and enough protective cover against avian predators, and were the habitat with the highest percentage of green vegetation cover (80-90%) from spring to autumn, providing mammals with year-round protein-rich green food. However, field margins were the most stable habitat, with small mammal densities about 2.3 times higher on average than within fields and 8-9 times higher than in cereal fields in spring and autumn when soils are tilled and bare), and provide a key refuge in winter and spring. These small mammals are key prey items for birds of prey that use arable farmland for hunting in Spain, such as Montagu’s Harrier, Hen Harrier and Black-winged Kite.

Montagu’s Harriers also forage over alfalfa sown on fallow fields in the Netherlands intensively (Schlaich et al, 2015), preferring mown to unmown strips, as voles were more easily caught on the mown strips. Vole abundance was at least five times higher on the fallow fields, which were sown half with alfalfa and half with a cereal-grass-flower seed mix designed to benefit small mammals, than in other arable crops.

Hamster (Cricetus cricetus)
The European Hamster is closely associated with arable farmland on deep well drained soils such as loess soils. Hamsters are therefore typical inhabitants of highly arable farmland areas. Their distribution in France (O’Brien, 2015) and Germany (Albert, Reiners and Encarnação, 2011) has been found to be related to the cultivation of alfalfa in cereal rotations. However, a German study found that hamsters were much more abundant on wild-flower sown fallow than any other habitat type within arable farmland (Fischer and Wagner, 2016). The fallow was sown with a diverse flower-rich seed mix and then left undisturbed for five years.

2.4.11 Creation of "buffer zones" for high nature value areas, Natura 2000 or other biodiversity protection sites, including along hedgerows and water courses

See evidence for biodiversity impact of hedges, field margins and buffer strips above.

2.4.12 Borders, in-field strips and patches managed for wildlife or specific fauna (herbaceous border, protection of nests, wildflower strips, local seed mix, unharvested crops)

See evidence for biodiversity impacts of sown field margins, cover crops sown for wildlife and wild bird seed mixes on fallow above.
2.4.13 PRODUCTION ON ARABLE LAND WITH NO USE OF FERTILISER (MINERAL FERTILISER AND MANURE) AND/OR PLANT PROTECTION PRODUCTS, AND NOT IRRIGATED, NOT SOWN WITH THE SAME CROP TWO YEARS IN A ROW AND ON A FIXED PLACE

A field assessment in Germany that compared various possible EFA options found the highest biodiversity, soil, water and climate benefits for extensively managed cereal-flowering plant mixes (e.g. wheat/camelina, oats and arable weed seed mix) and lentils. All the options were managed with no application of pesticides and fertilisers, no soil cultivation before 30 November, and no cutting between 15 April and 1 August (Dziewiasty et al, 2013). See also evidence of biodiversity impacts of green manure nitrogen fixing crops such as alfalfa and of crop diversity in low intensity arable systems above.

2.4.14 CONVERSION OF ARABLE LAND INTO PERMANENT GRASSLAND EXTENSIVELY USED

See evidence of biodiversity impacts of permanent grassland above.

2.4.15 LANDSCAPE STRUCTURE COMBINING EFAS IN AN ECOLOGICAL NETWORK

Member States can choose to offer agri-environment schemes that fund added management on EFAs that provide biodiversity and environmental benefits, including the integrated planning of EFAs with other farm habitats and the creation of new farm habitats to connect all the habitats across the farm and with the surrounding landscape.

A study in southern England shows the scale of loss of semi-natural habitats and connectivity in the farmland landscape which needs to be counteracted. The study assessed landscape change over 70 years on a large area (2500 km²), at high resolution (25 x 25 m) in southern England, and concluded that the connectivity between fragments had dropped drastically, by up to 98%, and 97% of the semi-natural grasslands had been lost to intensification (Hooftman and Bullock, 2012). Even though the EFAs cover a small proportion of farmland, a real improvement in habitat value can have a significant impact on species richness.

A study which modelled the effects of the density of natural elements in agricultural landscapes on species richness of vascular plants, breeding birds, butterflies, hoverflies, dragonflies, and grasshoppers at the national scale (Cormont et al, 2016) found that even landscapes with 3–7% of natural elements harboured generally 37–75% of maximum species richness, but the shape of the species richness response differed between landscape types for butterflies. The study recommended the development of tailor-made guidelines at regional levels to ensure that EFA habitats increase species richness.

A study modelled the effects of the cultivation of peas and beans, permanent and rotational set-asides, permanent extensive grasslands, and herbaceous and woody field margins on European Brown Hare (Lepus europaeus) in Danish farming landscapes characterised by low hare densities (Langhammer et al, 2017). An increase in the area of any of the options to 5% of the landscape had significant positive effects on hare populations, but only the permanent set-aside scenario led to densities of more than 10 females per km², the threshold for population viability, in all landscape configurations, whilst herbaceous field margins had the same effect in two of the three landscapes. As EFA in Denmark will not result in such a large increase in any one of these EFA options, because farmers can choose from a range of options, and are not required to maintain multi-year fallow, the study concludes that the EFA requirement is currently insufficient to improve the living conditions of the brown hare to a necessary degree.

Several studies and expert consultations have attempted to compare the relative potential of the EFA options for environmental benefits, and come to contrasting conclusions depending on whether only biodiversity is considered or other environmental benefits are also evaluated:

- A survey of 88 ecologists in 17 European countries (Pe’er et al, 2016) scored the EFA options for biodiversity in the following order: fallow > buffer strips > landscape features > nitrogen fixing crops > catch crops / green cover > agroforestry > short rotation coppice > afforested areas.
The EFA calculator ranked EFA options for potential benefit with respect to biodiversity, ecosystem services, and ease of management, for a series of regional case studies across the EU, in the following order: terraces > afforested areas and tree groups > stone walls > buffer strips > tree lines > fallow > ditches > ponds > single trees > hedges & wooded strips > field margins > catch crops > N-fixing crops > SRC > agroforestry (Figure 5.3 in Tzilivakis et al, 2015).
3 LITERATURE REVIEW ON THE EFFECTS ON SOILS OF FARMING PRACTICES RELEVANT TO THE PILLAR 1 GREENING MEASURES

3.1 INTRODUCTION

Soil is defined as the top layer of the earth’s crust and is composed of mineral particles, water, air and organic matter, including living organisms. It is a complex, mutable, living resource which performs many vital functions: food and other biomass production, storage, filtration and transformation of substances including water, carbon and nitrogen. Soil further serves as a habitat and a gene pool, and provides a basis for human activities, landscape and heritage, and the supply of raw materials (Louwagie, Gay and Burrell, 2009b).

Agriculture occupies a substantial proportion of European land and has a major impact on soil. The EU’s ‘Sustainable Agriculture and Soil Conservation through simplified cultivation techniques’ (SoCo) project identified six soil degradation processes that are closely linked to agriculture (Louwagie, Gay and Burrell, 2009b):

- Erosion – by water, wind and tillage
- Decline of soil organic carbon
- Compaction
- Salinisation and sodification
- Contamination
- Declining soil biodiversity

This literature review summarises the evidence for the benefits of the full range of greening measures and equivalent practices on soils within the European Union in ameliorating these soil degradation processes. Decline of soil organic carbon and emissions of other greenhouse gases are covered in more detail in the literature review covering greenhouse gas emissions, and this soil review cross refers to it.

A feature of the greening measures is that many of them overlap, so, for example, crop diversification can be achieved through the use of Ecological Focus Area fallow and nitrogen-fixing crops. Because of this a certain amount of cross referencing is inevitable.

3.2 CROP DIVERSIFICATION GREENING MEASURE

It is not always easy to distinguish the benefits of crop diversification from crop rotation, since diversification normally involves rotation. Because the greening measure also allows the inclusion of fallow and of nitrogen-fixing crops, the benefits of this measure are also sometimes difficult to distinguish from those with which it overlaps.

3.2.1 DIVERSIFICATION OF CROPS

In contrast to crop rotation, there is comparatively little literature on the impacts of crop diversification on soils in a European context. Crop diversification will usually lead to crop rotation, so the impacts on soils might be expected to be similar, though at the time crop diversification was proposed as a greening measure, a consortium of NGOs were amongst those arguing that this was not necessarily a safe assumption (PAN et al, 2012). They pointed out that, under the greening rules then being proposed, a farmer with 100 hectares of land could plant 70 hectares of maize, 25 hectares of wheat and five hectares of barley annually, and repeat that over subsequent years. They claimed that this type of diversification would not be enough to break monocultures, or result in the agronomic and environmental benefits of rotations in annual cropping systems. This claim appears to receive some support by the advice to farmers in the UK from the Farm Advice Service, which in 2015 put out a fact sheet (FAS, 2015) that advised farmers that were currently block cropping wheat that they could still grow efficiently whilst meeting the crop diversification requirements by planting 75 per cent winter wheat, 20 per cent autumn sown spring wheat and five percent temporary grass
or fallow. In such a rotation, any benefit to soils would arise only from the introduction of an area of fallow. The benefits for soil of fallow are covered under the EFA section of this review.

Despite there being no obligation to rotate crops, anecdotal evidence suggests that many farmers are choosing to rotate crops for purely agronomic reasons. Where this happens the benefits or otherwise for soils will be as described in the following section on crop rotation.

### 3.2.2 Crop Rotations

Crop rotation is a planned system of growing different kinds of crop, in the same sequence, on the same land and over three or more years (Louwagie, Gay and Burrell, 2009a). In the EU, crop rotations typically last from three to five years in conventional agriculture and from five to ten years in organic agriculture (BIO Intelligence Service, 2010).

A number of studies have shown that crop rotations can benefit soils. A good example of these was a study carried out in the United States (Tiemann et al, 2015) which showed that increasing the number of crop species grown in a three year rotation from one to five increased the quality and quantity of crop residues returned to the soil. This led to increased soil organic carbon, increased microbial activity, nitrogen content and, in the most diverse rotations, an increase in soil fungi relative to bacteria, which helped develop a more stable soil structure.

A review of the literature on the impacts of agricultural management on soil biota (van Eekeren, Murray and Smeding, 2007) concluded that a ley/arable crop rotation would be better for soil biota, and hence for a range of soil ecosystem services, than continuous arable. The review recommended the use of grass/clover mixes in the ley phase to increase earthworm populations and hence soil structure.

A study in central Spain (Boellstorff, 2005) looked at the impact on soils of replacing a three year winter wheat, pasture legumes and fallow rotation with a longer rotation in which five year periods of management as pasture alternate with five years of wheat alternating with fallow. This study predicted that making this change would increase soil organic carbon by 1.3 times over 25 years with little overall loss of production. Increasing levels of humus were also likely to improve soil structure and fertility.

A review (BIO Intelligence Service, 2010) identified the key factors that influenced the impact of crop rotations on soil organic matter, soil structure and soil erosion:

- For soil organic matter, one key factor in determining the impact of crop rotation is the amount of biomass returned to the soil (whether as unharvested above ground biomass, roots or extra-root materials). Crops with a high yield and crops leaving high amounts of root material will tend to return more biomass to the soil.

- Another key factor for soil organic matter is the amount and nature of tillage involved in managing the different crops in the rotation. Generally speaking, conventional inversion tillage reduces soil organic matter compared to no tillage.

- The overall impact of a crop rotation on soil organic matter will depend on the interaction of these two factors with a range of other factors including the farming system (arable or mixed), the local conditions, whether or not catch and cover crops are used, whether legumes are included in the rotation and whether organic manures are used. The review concluded that long rotations including grasslands were the most likely to maintain soil organic matter.

- The key factor in determining soil structure is its porosity. Soil pores are essential for many soil functions so an increase in porosity (measured by a reduction in bulk density) generally implies an improved soil structure. As well as the actual crops grown, factors affecting porosity are tillage, traffic and organic fertilisation. Cultivation produces a short term increase in porosity but, over the medium to long term, no till often produces an increase in porosity. The movement of farm machinery, especially in wet weather, is a major cause of compaction.
Organic fertilisation generally increases porosity. Attempts to measure the impact of different rotations on soil structure have often been inconclusive, but it seems likely that rotations that include temporary grasslands are the most likely to be beneficial, which is consistent with the recommendation of van Eekeren et al.

- The impact of crop rotations on soil erosion will depend on the extent to which the individual crops leave soil vulnerable to erosion events. As a general rule, row crops such as maize, sunflowers, sugar beet and vegetables offer little protection against erosion whereas cereals and rapeseed, which generally produce ground cover through the winter months are better. Other factors include the nature and timing of tillage and the extent to which crop residues are left on the surface. Breaking up the soil surface to increase roughness can reduce soil erosion, but no till can also reduce it by improving soil structure. Crops leaving low crop residues, such as potatoes or peas, can increase the risk of soil erosion.

Overall the review concludes that diversified and long crop rotations offer the widest range of environmental benefits. It also concludes that soil function will particularly benefit from rotations which:
- Match the choice of crops to the quality of the soils
- Include catch and cover crops
- Balance crops with lower and higher nutrient needs
- Include legumes (except in areas of manure surplus)
- Include temporary grasslands.

The Joint Research Council’s Sustainable Agriculture and Soil Conservation (SoCo) project also reviewed the impacts on soil of crop rotations and concluded that they can improve soil quality and soil fertility, decrease soil erosion and increase the recycling of mineral elements. They concluded that these benefits arose particularly from the inclusion of grass and legumes in the rotation (Louwagie, Gay and Burrell, 2009a). A more general hierarchy of agronomical criteria necessary to build an effective crop rotation (Viaux, 1999), was included in the SoCo report. These criteria, starting with the most significant, are as follows:
- List the species most adapted to the area
- Introduce the maximum number of families and species to the rotation
- Introduce at least one legume to the rotation
- Incorporate at least a third of the cereal straw
- At least every three years introduce a long intercrop, integrating spring and winter cropping
- Sow nitrogen demanding winter crops after annual legume crops
- Alternate phosphorus and potash-demanding crops with those having a low demand for these elements

Overall, the literature suggests that, whilst crop rotations can benefit soils, the extent to which they do depends on the crops that make up the rotation and the way they are managed. It cannot be assumed that soils will automatically benefit from crop rotation. The most beneficial rotations for soils appear to be those that involve many different crops, which include at least one leguminous crop and which include grassland. One example comes from Martin-Rueda et al (2007), who found that a two crop rotation with barley had no effect on SOC levels.

3.2.3 Winter Soil Cover

Leaving crop residues in place over winter can help reduce soil erosion. Some evidence relating to this can be found in the crop rotation and the cover crop sections.

3.2.4 Catch crops

The evidence is presented in the EFA section.
3.3 Maintenance of permanent grassland greening measure

There is an extensive literature demonstrating that soils under grassland tend to be better than those under arable cropping, but that the way grasslands are managed can have a major impact on soil quality. The greenhouse gas emissions literature review also covers the evidence on the effects of grassland retention and different forms of grassland management on soil carbon.

3.3.1 No ploughing of permanent grassland

The quality of soils, measured by the standard indicators of organic carbon and aggregate stability, has been shown to be higher in permanent grasslands than in arable (Gardi et al, 2002), a conclusion supported for soil structure by a separate study in the Veneto region of Italy (Giardini and Morari, 2004). A study of permanent grasslands in northern France (Plascart et al, 2008) found much richer fungal and bacterial microbial communities in permanent pasture than in adjacent arable fields and a strong relationship between fungal genetic diversity and the age of the grassland. The SoCo project (Louwagie, Gay and Burrell, 2009a) also found data showing that across the EU permanent pasture had the lowest mean value of soil erosion risk (less than 0.5 tonnes per hectare per year) of any form of agricultural land use. This implies that it is particularly important to retain permanent pasture in situations where soils are at high risk of erosion, such as on steeper slopes.

The EU funded Climsoil project (Schils et al, 2008) reviewed much of the evidence on climate–related aspects of soil management. The project concluded that, when soils are compared on a like-for-like basis, soil organic matter levels are always higher in soils under permanent grassland than in identical soils under arable land. The same report also found that whereas grassland soils generally accumulate carbon, cropland soils generally act as a carbon source, though quantification is difficult and different studies produced very different estimates.

The benefits of maintaining permanent pasture to retain soil carbon varies according to the soil type and the geographic location. Topsoil organic carbon content is generally higher in permanent pastures in northern Europe than in southern Europe (Gobin et al, 2011). It is particularly high in peat soils.

The most widespread practice causing damage to peat soils is drainage, with around 16 per cent of the overall European peatland area currently used for agricultural purposes (both for cropland and grassland management) and presumably drained, including the vast majority of peats in continental western Europe (Byrne et al, 2004). However, when peat soils are both drained and cultivated they become very vulnerable and can release large quantities of soil carbon through oxidation, wind and water erosion. A combination of shrinkage, compression, oxidation, wind erosion, soil export on crops and accidental burning are estimated to have reduced ground levels in parts of the cultivated fens of eastern England by as much as 4.9 metres since drainage began (Holman, 2009). CO₂ emissions from drained peats in East Anglia are recognised as one of the largest land use related sources of CO₂ in the UK. Estimates of CO₂ loss range from 109 grams of carbon per square metre per year to 1280 grams of carbon per square metre per year for peat depths of less and greater than one metre, respectively (Baggott et al, 2007). The proportion of peatlands under cultivation across the EU varies markedly between the countries of the EU. In some regions of some countries, the proportion of peat soils under cultivation is extremely high (over 73 per cent), such as in parts of the UK and France (Gobin et al, 2011).

Whilst greening imposes some restrictions on the ploughing of permanent grasslands, it does not control how they are managed. This affects all aspects of soil quality. The climate change literature review presents a range of evidence on the impact on the soil carbon of permanent grasslands under different grazing intensities, increased productivity, nutrient management and grass species diversity. The evidence presented suggests that carbon sequestration decreases with grazing intensity, but the evidence on the impact of nutrients seems more complex. Under certain circumstances, adding fertiliser or irrigating can increase carbon sequestration, but adding nutrients to previously nutrient-poor grasslands can lead to large losses of soil carbon. Introducing new deep-rooted grass species to low productivity grassland swards can increase carbon sequestration. One recent study (Ward et al, 2016) has found that, when soil carbon in the top one metre of soil was measured, Soil carbon stocks were higher in immediately managed permanent grassland than in either extensively managed or intensively managed permanent grassland. Another study of cattle grazed grasslands in northern Spain (Estavillo et al, 2002) found that adding fertiliser to grasslands produced an increase in the...
rates of \( \text{N}_2\text{O} \) production, \( \text{N}_2\text{O} \) production from nitrification, \( \text{N}_2\text{O} \) production from denitrification and total denitrification (\( \text{N}_2\text{O} + \text{N}_2 \)).

The avoidance of overgrazing has been identified as a key factor in reducing the risk of soil erosion and loss of soil organic matter from permanent grasslands (Poláková, Hart and Black, 2012). There is considerable variation between the robustness of different soil types in different geographic locations to grazing management. Whereas on average grazing levels of up to 1.5 livestock units per hectare were found to be optimal for carbon retention in UK permanent grasslands (Ward et al, 2016) on upland organic soils levels in the UK the threshold above which land becomes vulnerable to erosion can be as low as 0.55 sheep per hectare (0.08 livestock units per hectare) (Rawes and Hobbs, 1979). The Pontbren project in Wales (Keenleyside, 2012) found evidence of high levels of surface run off from intensively managed permanent grassland in Wales as a result of the compaction of the surface layers of soil by sheep.

A wider review of the impacts of grassland management on soil functionality (Rumpel et al, 2015) found evidence that although grasslands are in general favourable systems in terms of soil organic matter storage and nutrient cycling, intensification of grassland management may decouple elemental cycles and favour losses of reactive nitrogen and phosphorus. The same review found evidence that grazing may be more beneficial for soil organic matter content compared to mowing up to a certain animal density, depending on soil type and pedoclimatic context.

This review also looked at the effect of the botanical composition of grassland swards. It found evidence that botanically diverse swards penetrate larger volumes of soil due to diverse and complementary root traits. Due to this increased root activity, and in turn higher soil organic carbon and soil microbial biomass concentrations, aggregate stability (a measure of soil porosity) increases. The introduction of forage legumes generally has a favourable impact on soil due to their deep rooting system and easily decomposable organic residues rich in nitrogen, which promote the development of micro-fauna and soil microflora. The direct availability of the fixed nitrogen to the roots also minimizes nitrogen losses. Combining grasses and legumes in the sward can increase the efficiency of nitrogen fixation. The review concludes that further progress is necessary in order to optimise grassland management strategies through species choice, fertilisation, and grazing intensity for intensively used grassland systems.

The evidence reviewed suggests that, to the extent that greening has prevented the ploughing of permanent grasslands, it will have delivered a range of benefits for soils. However, particularly in the case of the low productivity botanically diverse permanent grasslands most likely to be identified as ESPG, preventing ploughing without controlling the intensification of grassland management may lessen but will not prevent soil damage.

Verbruggen et al (2010) compared agricultural land with grassland that was mown twice a year and used for irregular grazing. They found that the average number of arbuscular mycorrhizal fungi taxa was highest in grasslands (8.8), intermediate in organically managed fields (6.4) and significantly lower in conventionally managed fields (3.9).

### 3.3.2 Reseeding permanent grassland (including with and without ploughing)

Ploughing and reseeding permanent pastures has some adverse effects on soils. A study in Ireland (Necpálová, Casey and Humphreys, 2013) found that the ploughing and reseeding of permanent grassland on a dairy farm led to the loss of 3.1 tonnes per hectare of total nitrogen and 13.4 kilogrammes of soluble nitrogen in the 1.7 years after ploughing. Another study on cattle grazed permanent pasture in northern Spain (Estavillo et al, 2002) found that ploughing promoted soil organic nitrogen mineralisation, which led to an increase in the rates of \( \text{N}_2\text{O} \) production, \( \text{N}_2\text{O} \) production from nitrification, \( \text{N}_2\text{O} \) production from denitrification and total denitrification. These effects were greater when fertiliser was also applied.

A study in the Netherlands (Vellinga, van den Pol-van Dasselar and Kuikman, 2004) found that that ploughing permanent grassland for conversion to ley rotations and for grassland renovation was responsible for considerably increased \( \text{N}_2\text{O} \) and \( \text{CO}_2 \) emissions. Although soil carbon and nitrogen decreased sharply during the brief fallow periods associated with grassland renovation, the emissions from grassland renovation were much less than from conversion to arable. The emissions of \( \text{N}_2\text{O} \) could be further reduced by undertaking renovation in the spring.
The study of permanent grasslands and restored grasslands in northern France referred to earlier (Plassart et al, 2008) not only found much richer fungal and bacterial microbial communities in the soils of permanent pasture than in adjacent arable fields, but also found that soil fungal biomass recovers quickly after reseeding. There was however a strong relationship between fungal genetic diversity in the soil and the age of the grassland, with the fungal community continuing to develop for more than ten years. The study pointed out soil fungal communities are important as they can decompose recalcitrant organic matter, contribute to the soil structure, to moisture retention and to the resistance of soils to erosion.

The evidence for the impacts on soil biota of different forms of grassland management was reviewed in 2007 (van Eekeren, Murray and Smeding, 2007). This found that the microbial communities of permanent grassland soils may take decades to fully recover. Earthworm populations, which have a major impact on soil structure, particularly drainage, are likely to take at least five years to recover. The authors of the review concluded the results they had found showed that it takes years to restore the soil food web and its services in a grass ley to the level of permanent pasture. They recommended that when the plant-soil cycle in a permanent pasture is running smoothly, ploughing up a permanent pasture for ley-farming should be avoided.

The greenhouse gas emissions literature review presents some evidence (Tilman, Hill and Lehman, 2006) that introducing new deep-rooted grass species can increase soil carbon, particularly on low-productivity pastures.

Direct evidence on the comparative merits of grassland renovation with and without ploughing is hard to find, but one study in Wales (Roberts, Hudson and Roberts, 1989) found that there were losses of nutrients, particularly nitrogen, when upland permanent grasslands were disc harrowed and reseeded, but that these losses were reduced, especially for nitrogen, when reseeding was done using a minimum cultivation technique. There is a lot of evidence that reduced cultivation has a range of benefits for soils under arable crops (BIO Intelligence Service, 2010; Busari et al, 2015), so it would seem likely that reseeding using minimum cultivation will have similar benefits. The establishment of re-seeded swards without full cultivation can however be difficult (van Eekeren, Murray and Smeding, 2007).

The evidence reviewed would seem to suggest that the regular reseeding of permanent grassland, particularly where that involves ploughing, will reduce the benefits to soils of retaining permanent grassland. Soils under such management are however still likely to be better than those under continual cropping. Management techniques such as undertaking grassland renovation in the spring, re-seeding with grass/clover mixes and using minimum tillage techniques where practical can limit the adverse effect of re-seeding on grassland soils.

### 3.3.3 Ploughing in one area and converting land to PG in another

Section 1.3.2 presented a range of evidence showing that the ploughing and re-seeding of permanent grassland releases nutrients, particularly nitrogen, from the soil, increases the emissions of greenhouse gases (CO₂ and N₂O) and adversely affects soil biota. The study from the Netherlands previously cited (Vellinga, van den Pol-van Dasselar and Kuikman, 2004) found that there were large releases of greenhouse gases when permanent grassland was converted to arable (250 tonnes of CO₂ equivalent per hectare). Losses were smaller when permanent grassland was converted to a ley/arable rotation (150 tonnes of CO₂ equivalent per hectare) but still larger than that for grassland renovation. Emissions were very high immediately after ploughing and decreased slowly over at least 50 years.

The study of permanent grasslands and restored grasslands in northern France (Plassart et al, 2008) also found long-lasting effects of ploughing, with soil fungal communities taking more than ten years to recover. The review of the impacts of management on grassland soil biota (van Eekeren, Murray and Smeding, 2007) found that the ploughing up of permanent grassland had long term impacts on different aspects of soil biota. Earthworm populations took more than five years to fully recover, nematode populations more than ten years and microbial communities could take decades.

Despite this, the 2007 review concluded that the soil biota of ley/arable rotations were likely to be better than those of soils subject to continuous cropping. It also suggested that soil biota, and hence other soil properties, could be improved by adding clover to the grass leys. The BIO Intelligence
Literature reviews on the effects of farming practices associated with the CAP greening measures on climate and the environment

Service Review cited in section 1.2.2 (BIO Intelligence Service, 2010) also suggested including temporary grasslands in a rotation to improve soil organic matter.

A review carried out for the UK government (Wiltshire et al, 2014) found that introducing grass leys to continuously cropped land had a range of benefits for soils, with a decreased risk of erosion, increased porosity and structural stability and an increased supply of biologically fixed nitrogen (where legumes were used as part of the ley). There was some increase in soil carbon, particularly in longer grass leys, but ley-arable grassland was up to ten times less effective at storing C than permanent grassland. This was because most of the carbon sequestered by the ley was released again during the period of arable cropping.

The evidence reviewed suggests that, compared to maintaining existing permanent pasture, ploughing in one area and converting land to permanent grassland in another has a range of deleterious impacts on soils, some of which can be long lasting. However, compared to continuous cropping, the use of grass leys, especially leys that are left in place for some time, can benefit soils.

3.4 ECOLOGICAL FOCUS AREAS

3.4.1 LAND LYING FALLOW

There is a considerable literature on the effects of fallow on soils, much of it relating to the former CAP requirement for set-aside. From this it emerges that the management of fallow is one of main factors determining the effect it has on soil.

The degree to which the soil is covered is the key factor in relation to soil erosion. A modelling study (Panagos et al, 2015a) recognised three categories of fallow aside when modelling the impact of soil cover on soil erosion:

- bare land with no crops
- land with spontaneous vegetation
- land sown for the production of green manure

Soil erosion cover factor (C-factor) is one of five factors used to assess the risk of soil erosion in the Revised Universal Soil Loss Equation (RUSLE). This study assigned bare land with no crops and with down slope drainage the C-factor reference value of 1.0 and the average of the three categories was given a value of 0.5.

Differences in the speed and extent to which fallow land develops a vegetation cover seem to largely explain the different effects that fallow has had on soil erosion in different studies. A review of the evidence for the benefits of set aside found that unseeded fallow (IEEP, 2008) generally increased the soil erosion risk whereas fallow sown with a cover crop generally reduced the risk. The review concluded that non-rotational set-aside with a grass cover was the best option for areas of high soil erosion risk.

A study of soil erosion on set-aside in central Spain (Boellstorff and Benito, 2005) found that in that area set aside encouraged the use of unseeded fallow. This did not quickly develop vegetation cover and led to an increase in the area of land at high erosion risk. By contrast a study of soil erosion on sandy soils in east Shropshire, UK (Fullen, 1997), found that set-aside with a grass cover was very effective at limiting soil erosion even on slopes as steep as 15°. Soil erosion rates were high (an average of 0.82 tonnes per hectare per year) immediately after falling but fell to tolerable levels once vegetation cover reached 30%. Under established grass swards erosion rates were in the range 0.1 to 0.5 tonnes per hectare per year.

The effect of fallow on soil organic matter also appears to be closely related to the extent and nature of the vegetation that develops. The IEEP review cited above concluded that green cover on set-aside increased soil organic matter and improved soil structure. In the east Shropshire study cited above soil organic matter content declined when the soil was bare, but increased by an average of 0.39 per cent after two years of grass covered fallow and by an average of 0.78 per cent after four years. A modelling exercise looking at the potential impact of grass fallow in south east England and Sweden (Sahrbacher et al, 2015) predicted that planting miscanthus (Miscanthus giganteus) for bioenergy on fallow could result in a faster accumulation of soil organic matter (1.5 per cent per year rather than 0.5 per cent per year) than would be obtained from a normal grass fallow.
A study of the effects of set-aside management on the husbandry and yield of subsequent cereal crops in the UK (Smith et al, 2000) compared the effects of set aside left to regenerate naturally with those of set-aside sown with perennial rye grass (*Lolium perenne*) and set-aside sown with a mixture of rye grass and white clover (*Trifolium repens*). The study found that most of the differences in yield observed between the three treatments were due to differences in soil mineral nitrogen content. This was highest in the plots sown with clover in the seed mix and the cereal crops subsequently grown on these plots showed enhanced tiller production and improved grain yield and quality, again suggesting that the vegetation cover is the most important factor in determining the effect of fallow on soil nutrient status.

The Highfield experiment at Rothamsted in England provided a unique opportunity to compare the soils of initially similar plots that have been in continuous bare fallow, continuous grassland and continuous arable for more than 60 years (Wu et al, 2010). Total carbon and nitrogen concentrations in fallow plots were found to be lower than in either the continuous grassland or continuous arable. There was 82 per cent less carbon in the fallow than in the grassland plots compared to 70 per cent less in the arable plots. Despite this the diversity, though not the numbers, of soil micro-organisms was similar across all three treatments, suggesting that communities of soil microorganisms may have greater potential to recover than previously recognised.

A study from the Castro Verde region of southern Portugal (Castro et al, 2016) found that there were differences between the soil microbial communities of the wheat and fallow phases of a four year cereal/fallow rotation. The main changes in the structure of microbial communities happened between the first and second year of fallow, probably due to changes in the quality and quantity of litter inputs. The impact of the fallow stage on microbial communities was complex: bacterial diversity was found to be higher during the wheat stage while fungal diversity was either higher or unchanged during the fallow.

A number of other studies across Europe have looked at the effects of fallow on soil organisms. A study done in Hungary (Tóth et al, 2016) compared the macro-decomposer arthropods (isopods and millipedes) found in fields that had been set aside for one, two and three years with those found in wheat fields and semi-natural grasslands. The isopods were more abundant and diverse in the set-aside plots than in the wheat, and became more abundant with the age of the set-aside. Abundance and diversity again appeared to be related to the cover and diversity of plant growth. The three year old set-aside plots had more species rich collections of millipedes than the semi-natural grassland plots.

A study of the effects of set-aside on earthworms in four European countries, The Netherlands, Sweden, The Czech Republic and the UK (Gormsen et al, 2004), found rather different results in the different countries. Across all four countries the effect on earthworms of three different set-aside management regimes (high diversity seed mix, low diversity seed mix and natural colonisation) were compared with that of continued arable cropping. In this study there was little difference between the treatments, but the numbers and biomass of earthworms were significantly higher in the set-aside than in the arable in two out of the four countries, though not in the other two. There were however some effects of vegetation cover, in Sweden earthworm biomass was positively related to legume biomass (confirming the results of other studies cited in this review) and in England to grass biomass.

As with ley grasslands, there can be adverse impacts on soil when they are ploughed up. The IEEP review found evidence that reverting set-aside back to arable increased N₂O emissions from soils. The modelling exercise looking at the potential impact of grass fallow in south east England and Sweden that was cited above (Sahrbacher et al, 2015) also predicted that, in order to reverse the decline in soil organic matter using just one year rotational set-aside, no less than 25 per cent of the arable area would need to be in set-aside at any one time.

The evidence presented here suggests that using fallow to help meet the EFA obligation has the potential to produce a range of benefits for soil, reducing erosion and increasing soil biodiversity, soil organic matter and soil nutrient. Most of these benefits are however dependent on the development of a green cover on the fallow land. Without an adequate cover fallow land can exacerbate erosion, reduce soil carbon and reduce soil nutrients. The benefits to soils of fallow also generally increase with the age of the fallow and, as with grass leys, there are adverse effects on soils when fallow is ploughed up.
3.4.2 Terraces

Terraces are created to stop or reduce soil erosion by intercepting surface run off, facilitating its infiltration or evapo-transpiration and/or channelling it at a controlled velocity so that erosion can be avoided. Most of the literature on terraces relates to their impact on soil erosion and this was reviewed by the SoCo project (Louwagie, Gay and Burrell, 2009a). This found good evidence that terracing can reduce run off and soil loss due to water erosion, especially when combined with other techniques such as contour ploughing, the maintenance of vegetation cover and strip cropping. However, it also found evidence that the neglect or abandonment of terraces can trigger large scale soil erosion. Attempts to establish new terraces have also sometimes led to the loss of existing soil and have sometimes proved unsustainable (García-Ruiz, 2010), possibly because their design has been influenced by considerations other than soil and water conservation, such as the ability to accommodate mechanised agriculture.

A comparative study of different soil and water conservation techniques in the Mediterranean region (Maetens, Poesen and Vanmaercke, 2012; Maetens et al, 2012) found that terracing was one of the more effective techniques for reducing soil loss and particularly effective at reducing run off and conserving water. There was however a lot of variability in the effectiveness of terracing between the sample sites.

This evidence suggests that well designed and maintained terraces can be an effective method of conserving soil and water, but neglect can trigger large scale erosion. Construction of new terracing disrupts soils and needs careful design to ensure sustainability.

3.4.3 Landscape Features

3.4.3.1 Hedgerows and wooded strips

The UK NGO Hedgelink has summarised the benefits of hedgerows for soils (Hedgelink, 2017):

- Hedgerows can help prevent the loss of soil from fields, either through reducing wind erosion or through acting as a barrier to water-borne run-off. This is particularly so in arable areas, both where the land is flat and prone to wind-blow as in the Fens of East Anglia (UK) and in hilly areas where loss of soil following heavy rain can be a major problem especially in places prone to gully erosion.
- When planted along contours above ditches, streams or rivers hedgerows can play a valuable role in reducing the amount of polluting fertilisers, pesticides and sediment that reach watercourses, through acting as a physical barrier, through increasing infiltration into the ground, and through the trees, shrubs and other plants recycling nutrients.
- By decreasing wind speed in the boundary layer. In the Rhone valley in France a network of hedgerows has been planted to reduce water loss through evaporation in areas prone to drought.
- By helping to store water for later use. A 1ha field with a 50m hedgerow at the bottom can store between 150 and 375 cubic metres of water during rainy periods for slow release down slope, during dry periods. This effect is greatest in soils rich in clay or organic matter.
- By, together with woods, reducing the rate of flow of water within catchments, and so helping to reduce flooding downstream.

The EIP-AGRI focus group (EIP-AGRI Focus Group, 2016) looked at the benefits of landscape features, including hedgerows, for crop production. They identified a range of benefits including soil erosion control, nutrient retention and nutrient recycling. The group found evidence that hedgerows can take up nitrogen in the spring and recycle it as leaf litter. Levels of nitrogen were higher in areas of cultivated land adjacent to hedgerows, especially upslope, suggesting that the effect is due to a combination of recycling and retention. Although the group’s report shows hedges reducing soil loss, they recommend using grass buffers for this purpose in an arable landscape.

The effects of hedges in reducing soil erosion and increasing organic carbon were the subject of a study of the Bocage in the Amorican massif area of France (Walter et al, 2003), which has a dense
network of hedges built on stone walls, a form of boundary also found in south west England. This showed the soils under hedges accumulated large amount of organic material and had a low bulk density. The hedges had an anti-erosive effect, which also helped to retain soil carbon. As a result, a total of between 13 and 38 per cent of the total soil carbon in the Bocage was associated with hedgerows.

3.4.3.2 Isolated trees

There are unlikely to be more than local effects on soils from isolated trees.

3.4.3.3 Trees in a line

A farmer led project in Wales (UK), the Pontbren project in Wales (UK) (Keenleyside, 2012) has shown that shelterbelts, positioned across the slope in intensively grazed pasture can rapidly modify soil conditions and can substantially reduce the run off of water. The overall coverage of woodland in the project area was increased from 1.5 to five per cent by the establishment of a series of carefully sited, narrow, fenced shelterbelts running across the slope. The shelterbelts were planted with a mixture of mostly native trees and shrubs, fast growing pioneer trees, slower growing, longer lived trees and shrubs and small trees that could survive under and on the edges of the plantation and which could be coppiced. Existing woodlands were managed to provide long term continuity of cover.

The original impetus for establishing these shelterbelts was to provide shelter for livestock, but the farmers noticed they also absorbed run off during heavy rain. Their observations led to a major hydrological study of the Pontbren catchment. This showed that the soils under these shelterbelts had infiltration rates as much as 67 times greater than those in the adjacent intensively managed pasture. The rapid run off from the pasture land was mainly the result of soil compaction of the surface layers by sheep, but the beneficial effects of the shelterbelts could be only partly explained by the reduced compaction as a result of excluding sheep. Areas of fenced off grassland without trees had infiltration rates up to five times faster than under improved grassland. The majority of the increase was due to the greater porosity of soils under the trees. Tree roots were found to be directly creating channels for infiltration, with deep rooted species such as ash (*Fraxinus excelsior*) particularly effective in this respect.

The reduction in run off not only reduced soil erosion, but was shown to reduce the flood peaks in the stream draining the project area by up to 40 per cent. Data gathered from the Pontbren project has been used to refine mathematical models of the impact of land usage on catchment behaviour. This evidence shows that shelterbelts can, by the way they affect soils, be a highly effective way of reducing run-off on sloping ground. The success of the Pontbren project was however down to detailed planning by people who had an intimate knowledge of local conditions, and was in a landscape dominated by intensively managed grassland. To be effective elsewhere in different landscapes a similar level of site specific planning would be needed, though the mathematical models now available should be helpful.

3.4.3.4 Trees in groups and field copses

The effects on soils are likely to be similar to those described for lines of trees and for afforestation, but with more localised effects.

3.4.3.5 Field margins

See the section on buffer strips for impacts on soils

3.4.3.6 Ponds

Little literature has been found linking ponds with soil benefits. However Fiener, Auerswald and Weigand (2005) surveyed a number of small earth dammed detention ponds, created by raising the deepest part of field borders at the downslope end where runoff overtops. They found that the ponds were effective in: preventing linear erosion in downslope fields, trapping 50-80% of incoming sediment, reducing peak runoff rates and reducing peak concentrations of agrochemicals. The ponds were cheap to create and maintain, however regular draining by subsurface tile outlets is necessary to maintain the ponds and their effectiveness.
3.4.3.7 Ditches

No particular benefits for soils. The new or continued deep drainage of peat soils is extremely damaging as it causes the oxidation of peat. See the section on permanent grasslands for more details.

3.4.3.8 Traditional stone walls

The benefits for soils relate mainly to helping prevent the effects of soil erosion. Panagos et al (2015b) conducted a modelling study based upon the USLE/RUSLE support practice factor across the EU, and found that stone walls contributed 38% of total erosion risk reduction.

Stone walls perform a similar function as hedges - see the section on hedgerows for further evidence.

3.4.4 Buffer strips

The SoCo project (Louwagie, Gay and Burrell, 2009a) reviewed the evidence for the effectiveness of buffer strips in preventing soil erosion by wind and water and reducing the leaching of nutrients. The project found evidence that buffer strips situated next to arable land can significantly reduce the transport of suspended solids, nitrates and phosphates to watercourses. Buffer strips could abate 70 to 80 per cent of suspended solids, 70 to 98 per cent of phosphorus and 70 to 95 per cent of nitrogen. Their effectiveness does depend on the transport mechanisms by which these substances are reaching the watercourses, so they may be partially bypassed if a field is under-drained.

The SoCo project also found evidence that the effectiveness of buffer strips generally increases with their width. 12 and 24 metre wide grass buffer strips have been shown to a significant reduction in pollution, though the grass sward does require maintenance to maintain effective buffering.

A study in south western France (Furlan et al, 2012) modelled the effect of introducing grass buffer strips adjacent to water courses at two sites using the STREAM model. The modelling predicted that without the buffers a 40 mm spring rainfall with current cropping patterns would lead to 3116 cubic metres total water runoff and 335 tonnes of sediment yield at the first site and 3249 cubic metres and 241 tonnes at the second. Grass strips implementation could reduce runoff by about 40 per cent and sediment yield by about 50 per cent at the first site. At the second site, grass strips could reduce runoff and sediment yield by more than 50 percent, but changes in cropping pattern could reduce it almost totally. The authors pointed out that the grass strips along rivers and ditches would prevent soil sediments from entering the surface water but would not reduce soil losses. They also pointed out that there modelling showed that crop redistribution within the catchment was as efficient as planting grass strips (and would actually prevent erosion). They also made the point that the efficient management of erosive runoff requires coordination between all the farmers using the same watershed.

A review of the effectiveness of different soil conservation measures in a UK context (Wiltshire et al, 2014) also reviewed the evidence for the effectiveness of buffer strips in combatting soil erosion. This review also pointed out that buffer strips at the edges of field do not control soil erosion per se. Their function is to intercept run-off and any associated sediment before it leaves the field. It also found evidence that riparian strips should ideally be free-draining and have a good surface porosity to intercept surface run-off. The review presented evidence showing that the width of buffer strip required to prevent two tonnes per hectare per year of soil erosion varies enormously with soil and with the degree of vegetation cover. On clay soils with less than 20 per cent vegetation cover buffer strips would need to be more than 100 metres wide, whereas with more than 80 per cent cover the width required reduces to 24 metres. On medium soils with more than 80% vegetation cover six to eight metres width is required and for light soils with more than 80 per cent cover a two metre wide buffer can achieve this degree of protection. The UK review also found evidence that buffer strips could be highly effective in increasing soil organic carbon storage when introduced on arable land.

An experimental study done in the UK (Smith et al, 2008) looked at the effect of the management of buffer strips adjacent to arable fields on soil macrofauna. The experiment looked at the effects on soil macrofauna of sowing three different seed mixes and of three different management regimes, regular soil disturbance by scarification, spring cutting and herbicide application. The study found that diversity in the field margins was generally higher than in the crop, with the Lumbricidae, Isopoda and Coleoptera having significantly more species and/or higher abundances in the margins.
The plots that had scarified contained lower abundances and fewer species of isopod and reduced soil- and litter-feeder abundances and predator species densities, though these did recover later in the season. The authors concluded that to encourage the litter-dwelling invertebrates that tend to be missing from arable systems, the aim should be to minimize soil cultivation and develop a substantial surface litter layer. They pointed out that this would conflict with the management required to promote floristic diversity or to provide pollen and nectar resources on buffer strips.

A second study of soil macrofauna by the same group (Smith, Potts and Eggleton, 2008) showed that grass buffer strips also increased the species densities and abundances of earthworms, woodlice and staphylinid beetles. The biodiversity of the three main feeding groups – predators, soil ingesters and litter consumers – was also found to be significantly higher in the grass strips than in the field edges without strips, indicating that establishment of grassy margins in arable fields may enhance ecosystem services such as soil fertility and pest control.

The study by Panagos et al (2015b) also revealed that out of total P factor grass margins had the greatest soil erosion impact (57% of total erosion risk reduction).

The evidence reviewed suggests that the main benefit to soils of buffer strips is that of intercepting soil erosion and loss of mineral nutrients. A grass covering greatly increases their effectiveness, but to be fully effective the width of the buffer needs to be matched to the soil types and the siting of buffers needs to be coordinated at a catchment scale. Even so, buffer strips represent an ‘end of pipe’ response, intercepting, but not preventing soil erosion.

Other evidence reviewed suggests that buffer strips can locally enhance soil carbon storage and may have a wider benefit as reservoirs of soil biodiversity. There are some conflicts between the optimal management of buffer strips for soils, which requires a dense grass sward and a litter layer, and the optimal management for above ground biodiversity. This suggests that a variety of buffer strip management is probably required at farm scale.

3.4.5 AGROFORESTRY

Agroforestry is a collective name for land use systems where woody perennials are deliberately used on the same land management unit as either agricultural crops or livestock. These systems have a number of potential advantages including the ability to more effectively exploit soil resources due to the different rooting depths of the trees and the crop plants.

The SoCo project (Louwagie, Gay and Burrell, 2009a) reviewed a considerable volume of evidence for the impact on soils of silvoarable systems - those that combine trees or shrubs with arable cropping. The review found evidence of benefits from agroforestry for the maintenance of soil fertility, for soil erosion control, for increasing water holding capacity, for encouraging carbon sequestration and reducing nitrogen leaching. The reduction in nitrogen leaching was due to it being intercepted by tree roots growing at greater depth. The effects on soils were however found to be highly variable, depending on biophysical conditions, management, and the choice of crop types and tree or shrub species. More detail on the carbon sequestration benefits of agroforestry can be found in the greenhouse gas emissions literature review.

An experiment in North America (Anderson et al, 2009) demonstrated that introducing agroforestry to a watershed area previously under a maize/soybean rotation using agroforestry buffer strips, 4.5 metres wide and 36.5 metres apart significantly increased water infiltration and storage in the soil. Another North American study (Nair et al, 2007) looked at soils under range and pasturelands with and without tree growth in Florida. This concluded that the introduction of silvopastoral systems would enhance nutrient retention in the system and thus reduce the chances for nutrient transport to surface water. This was due to the more extensive rooting zones of the combined tree and forage system.

The SoCo review reports the results of a simulation showing the impacts on soils of a range of different combinations of tree and crop species at 19 different locations across the Atlantic and Mediterranean regions of Europe (Palma et al, 2007a). At each site, the effect of introducing agroforestry was examined at plot-scale by simulating the growth of one of five tree species at two tree densities (50 and 113 trees per hectare) in combination with up to five crops. At landscape-scale, the effect of introducing agroforestry on ten or 50 per cent of the agricultural area, on either the best or worst quality land, was examined. Across the 19 landscape test sites agroforestry had a positive
impact, with the strongest effects when introduced on the best quality land. The computer simulations showed that agroforestry could significantly reduce erosion by up to 65 per cent when combined with contouring practices at medium (between 0.5 and 3 tonnes per hectare per annum) and high (more than 3 tonnes per hectare per annum) erosion sites. Nitrogen leaching could be reduced by up to 28 per cent in areas where leaching is currently estimated high (more than 100 kg nitrogen per hectare per annum), but this was dependent on tree density. With agroforestry, predicted mean carbon sequestration through immobilization in trees, over a 60-year period, ranged from 0.1 to 3.0 tonnes of carbon per ha per year depending on tree species and location.

A meta-analysis of the benefits to biodiversity and ecosystem services of agroforestry in Europe (Torralba et al, 2016) which looked at 53 separate reports found that agroforestry enhanced erosion control and soil fertility, though there was a lot of variation between the studies. The authors concluded that both silvoarable and silvopastoral systems increased ecosystem service provision. They concluded that Mediterranean tree plantation systems should be especially targeted for agroforestry as soil erosion could be highly reduced while soil fertility was increased.

The evidence reviewed suggests that agroforestry has a range of potential benefits for soils, particularly when silvoarable systems are compared to conventional arable. A number of authors do however warn that the evidence suggests considerable variation in the outcomes and that designing and managing these systems to achieve optimal outcomes is complex.

3.4.6 STRIPS OF ELIGIBLE HECTARES ALONG FOREST EDGES

No specific evidence was found for the effects of this landscape feature on soils, but much of the evidence found for buffer strips would also apply.

3.4.7 SHORT ROTATION COPPICE

Most of the literature found for this review refers to the impacts on soils of willow (Salix spp.) and Poplar (Populus spp.) short rotation coppice, and to northern Europe. A number of studies have shown that in northern Europe short rotation coppice can enhance levels of soil organic carbon when established on formerly arable soils. One study in southern Germany (Díaz-Pinés et al, 2016) found that under hybrid poplar soil organic carbon increased by 400 kilogrammes per hectare over five years. Another study from north east Germany (Kahle et al, 2007) found an increase of 4 grams of carbon per kilogramme of soil after 12 years. A study of 14 long established short rotation coppice plots in southern Sweden (Dimitriou et al, 2012) found that organic carbon concentrations in the soils were significantly higher than in adjacent arable fields. Organic carbon in the topsoil was 9 per cent higher and in the subsoil it was 27 per cent higher.

The study in southern Germany (Díaz-Pinés et al, 2016) also looked at rates on N₂O emissions and nitrate leaching from hybrid poplar short rotation coppice. It found that rates of both N₂O emissions and nitrate leaching declined as the coppice became established. There was a brief increase in N₂O emissions after coppicing, but no significant increase in gross nitrification. Overall losses of nitrogen from soil decreased by 80 percent if no fertiliser was added. Nitrogen losses increased if 40 to 50 kilogrammes of nitrogen fertiliser per hectare were added, but were still 40 per cent less than those from adjacent arable land. The authors concluded that such additions of nitrogen were in any case not necessary, at least in the first cycle of coppice growth.

A second study of willow and poplar short rotation coppice in a drinking water catchment near Hamburg (Schmidt-Walter and Lamersdorf, 2012) also showed an increase in nitrogen leaching during the establishment phase, but lower levels of leaching once the coppice had established. The authors concluded that well managed willow and poplar short rotation coppice could be a useful alternative to arable cropping in drinking water catchments experiencing excessive nitrogen leaching from arable soils, though they expressed some concern about the stores of nitrogen building up in the soils under coppice. They found that if the coppice was left unharvested for long periods then rates of nitrate leaching began to rise again. They also expressed concern about possible releases of soluble nitrogen should the short rotation coppice be returned to arable cropping. Against the benefits of reduced nitrogen leaching they did find that willow coppice somewhat reduced groundwater recharge, but the effect was modest, providing the willow roots did not reach the water table.
A modelling study in north east Thuringia, Germany (Fürstenau and Barwölff, 2014) showed that short rotation coppice strips of 12 to 18 metres width, planted at a lower than normal density of 300 trees per hectare and harvested manually on a longer than normal ten year cycle had the potential to effectively buffer watercourses from soil erosion and nutrient run-off from arable land. The buffering effect was due to the permanent plant cover and the enhanced soil texture, leading to greater infiltration. The buffering effect would be temporarily reduced during and after harvest, but would be quickly restored.

Two studies found that short rotation coppice had some ability to remove heavy metals from contaminated soils. A German review of evidence (Baum et al, 2009) concluded that both willow and poplar can remove cadmium (Cd) and zinc (Zn) from contaminated soils, using their dense networks of fine roots, though these conclusions were based on pot-scale experiments. A field trial using willow in Sweden (Dimitriou et al, 2012) found evidence of cadmium uptake, as soils in long established short rotation coppice plots had levels that were 12 per cent lower than those in adjacent arable fields, but no difference was found in the levels of chromium (Cr), nickel (Ni) copper (Cu), lead (Pb) or zinc (Zn).

In one study, willow and poplar short rotation coppice was found to introduce ectomycorrhizal fungi to the soil and to increase the abundance of earthworms, but reduced the numbers of carabids. Another study found that overall soil biological activity, measured by the level of dehydrogenase activity, was higher under willow and poplar coppice. The bulk density of the soil decreased and porosity increased, indicating an improvement in soil structure. A study comparing fully a fully arable cultivation to arable with 20% SRC found that the SRC significantly sequestered carbon and reduced soil erosion (Langeveld et al, 2012).

The evidence reviewed here suggests that well managed willow and poplar short rotation coppice can benefit soil structure, increase soil carbon levels, reduce nitrogen leaching, reduce N2O emissions and act as effective riparian buffers, as well as perhaps having some potential to help reduce heavy metal contamination of soil. To maximise these benefits the coppice needs to be kept in place for as long as possible and ideally the coppice should be grown without the addition of nitrogen fertiliser, at least in the first rotation.

### 3.4.8 AFFORESTED AREAS

A study of the impacts of afforestation of former agricultural soils in three countries, Denmark, Sweden and The Netherlands (Rosenqvist, 2007) produced a ‘balance sheet’ of the advantages and disadvantages of afforestation for soils. One of the main advantages was that nitrate leaching was at least three to five times lower from the afforested areas than from arable land. There was also considerable carbon sequestration in both soils and above ground biomass. Against these advantages, water recharge was reduced and soils became more acidic. There was some variation depending on the soil type (heavy, nutrient rich soils were more likely to become nitrogen-saturated) and the tree species used (there was more nitrogen leached from soils under common oak (Quercus robur) than from soils under Norway spruce (Picea abies)).

A more widespread review of the effects of the afforestation of arable soils (Hansen, 2002) also concluded that afforestation does normally reduce nitrate leaching. It found that arable soils have a very open nitrogen cycle whereas that under mature woodland is normally more closed. However, there is considerable variation during the life cycle of the forest and between forests of different types. Nitrogen leaching is generally high during the establishment phase, especially if the soil is cultivated to aid tree establishment, when it can be higher than the normal rates from arable land. It then declines as the trees become established and increase their demand for nitrogen.

Once the canopy has closed, the demand for nitrogen from the trees reduces and the atmospheric deposition of nitrogen (dry and wet) increases leading to a risk that rates of leaching will start to rise again. The evidence on this was not conclusive, though the review did find evidence that there is a level of nitrogen leaching even from mature forests. It also found evidence that soils under afforested areas can take a long time (more than 100 years) to fully evolve from agricultural to woodland soils. Thinning has little impact on nitrate leaching, but clear-cutting does produce a large release of nitrate over a period of at least three years. Tree species such as alder (Alnus glutinosa) that fix nitrogen can produce high levels of nitrogen in soil water and forest streams and were not recommended for afforesting arable land.
This review also found evidence of an increase in soil carbon under afforestation and an increase in the carbon/nitrogen ratio. This will have profound effects on soil biota.

Afforestation is generally regarded as being a good way of reducing soil erosion, but a study in Portugal (Nunes, de Almeida and Coelho, 2011) found that, under Mediterranean conditions, there were very high rates of soil erosion from pine plantations, mainly due to the low levels of vegetation under the canopy. By contrast, the natural regeneration of shrub vegetation and oak forest did greatly reduce rates of soil erosion, improved levels of soil organic matter and improved soil structure.

There are a number of factors that need to be taken into account when deciding whether or not it will be beneficial to afforest agricultural land, and if so how to do it. The available information on the various factors such as the impact on nitrate leaching, carbon sequestration and water recharge have been used to build a decision support tool (Heil, Muys and Hansen, 2007).

The evidence reviewed suggests that afforestation can be very positive for soils and can have wider ecosystem benefits, but that care is needed in selecting the areas, the tree species and the management regimes. Using the wrong tree in the wrong place can be counterproductive.

3.4.9 Catch crops / Green cover

Catch or cover crops are crops sown in midsummer or autumn after harvest to provide green cover over winter and then destroyed or incorporated before the next commercial crop is grown.

A comprehensive review of soil protection measures applicable to the UK (Wiltshire et al, 2014) found good evidence that cover crops can reduce soil erosion, both by water and, where it is a problem, by wind. Cover crops with thick roots such as white mustard (Sinapis alba) and fodder radish (Raphanus sativus) were less effective than those with fine branching root systems such as rye grass (Lolium multiflorum) and rye (Secale cereale) at reducing soil erosion by water. Cover crops can reduce overwinter losses of nitrogen by 29 to 91 per cent compared to bare fallow and their repeated use can help to maintain levels of soil organic carbon, though occasional use has little impact. Cover crops are most effective on light soils as establishment can be difficult on heavy soils and the extra agricultural operations can cause soil compaction. Cover crops are also most effective when sown early in the season, so that a good vegetation cover can be established before the end of the growing season. Late sown crops can actually increase losses of nitrogen.

The SoCo project (Louwagie, Gay and Burrell, 2009b) found that cover crops were being used in two of its case study areas, the Bjerringo and Hvorsley area of Denmark and the West Vlaanderen area of Belgium. In the Belgian case study area (Verspecht et al, 2008) almost all farmers were using catch crops, mostly white mustard (Sinapis alba), phacelia (Phacelia tanacetifolia) and Italian ryegrass (Lolium multiflorum). Field trials in the area and the experience of the farmers interviewed, was that use of these cover crops substantially reduced soil erosion. The farmers also believed that the use of these cover crops increased soil organic matter, though field trials found that the effect was limited, which was consistent with the findings in the UK. The farmers interviewed were sufficiently convinced of the agronomic benefits of cover crops to continue planting them when a national subsidy scheme was withdrawn.

In the Danish case study area (Olsen and Dubgaard, 2008) farmers were obliged to use catch crops on part of their land to reduce nitrogen leaching and it had been found that they were also effective in preventing wind erosion on the light sandy soils found in the area. Beaudoin et al (2005) measured soil water and mineral nitrogen (SMN) three times each year on 36 sites with: wheat, sugar beet, pea, barley, and oilseed rape crops on a variety of soils over 8 years. The catch crops reduced the mean concentration of nitrate by 50% at the annual scale, and 23% at the rotation scale, the sugarbeet–wheat rotation gave the lowest concentration (38 mg L\(^{-1}\)) and the pea–wheat rotation the highest one (66 mg L\(^{-1}\)). The benefits of cover crops in reducing nitrate leaching was confirmed by a meta-analysis of trials comparing the rates of nitrate leaching and yields from cereals and bare fallow with those from cereals and non-leguminous cover crops (Tonitto, David and Drinkwater, 2006). Even though some fertiliser was used to help establish the cover crop, nitrate leaching was reduced by 70 per cent on average while the yields under non-legume cover crop management were not significantly different from those in the conventional, bare fallow systems.
A review of the impacts of cover crops in a North American context (Hartwig and Ammon, 2002) found a similar range of benefits to soil, primarily the conservation of soil nitrogen and a reduction in soil erosion. It also found that the use of nitrogen fixing cover crops could fix nitrogen for the subsequent crops without increasing over-winter losses. This was confirmed by the meta-analysis cited above (Tonitto, David and Drinkwater, 2006) which found that the use of unfertilised legume-based cover crops reduced nitrate leaching by an average of 40 per cent relative to conventional fertilizer-based systems.

A German study (Eisenhauer et al, 2009) found that legume cover crops provide high quality plant residues and root biomass for decomposers such as surface-feeding earthworms.

A review of the effectiveness of soil and water conservation techniques across Europe and the Mediterranean (Maetens, Poesen and Vanmaercke, 2012) found that cover crops were one of the more effective techniques for preventing soil erosion and conserving water (by reducing run off). Although a combination of mulching and buffer strips could achieve better control of soil erosion, cover crops were better at reducing run off, though there was a lot of variability in effectiveness between different studies. A modelling study of the impact of GAEC standards on soil erosion in Italy (Borrelli et al, 2016) predicted that an expansion in the use of cover crops from the current level of 15 per cent of cropland, combined with reduced tillage, could lead to an 8.5 per cent reduction in soil loss, though it would be insufficient to bring the very high rates of soil erosion in Italy down to the European average.

A Europe-wide review of measures intended to prevent soil erosion (Panagos et al, 2015a) found evidence from two separate studies that cover crops could reduce soil erosion by around 20 per cent. The review found that, while cover crops were extensively used in parts of Austria and in Belgium and the Netherlands they hardly used at all in Mediterranean countries. The review concluded that an expansion of cover crop use, together with greater retention of crop residues, would make a useful contribution to reducing soil erosion across Europe.

The evidence reviewed suggests that the major benefits of cover crops for soils are a reduction in soil erosion and reduced losses of nitrogen during the winter months. To be effective, cover crops have to be sown in time to establish a good cover in the autumn and they are most effective on lighter soils. Different cover crops have different benefits, with fine-rooted species being particularly effective at reducing soil erosion.

### 3.4.10 Nitrogen-fixing crops

Nitrogen-fixing crops grown as part of the EFA obligation can be divided into crops grown for their grain (e.g. field peas (Pisum sativum), field or fava beans (Vicia faba), lentils (Lens culinaris) and soybeans (Glycine max)) and crops grown as green forage and/or green manure (e.g. Lucerne/alfalfa (Medicago sativa), sainfoin (Onobrychis viciifolia), vetches (Vicia sativa), clovers (Trifolium spp.). If Member States make nitrogen fixing crops eligible on EFAs, they are required to select those relevant crops that contribute to improving biodiversity in their territory. Different EU countries have selected very different numbers of crops ranging from three in Northern Ireland (UK) to 20 in France (Underwood and Tucker, 2016).

Nitrogen fixing crops have a range of effects on soils. The most obvious of these is that they increase the amount of nitrogen in the soil. The mean value for the amount of nitrogen fixed per hectare across Europe in 2009 was 133 kilograms per hectare (Baddeley et al, 2013). Most of the nitrogen fixed by grain legumes is harvested and removed in the grain (Peoples et al, 2009), but the N benefit for following crops can often still be substantial, with several studies having demonstrated substantial savings (up to 100–200 kilograms of nitrogen per hectare) in the amount of nitrogen fertilizer required to maximize the yield of crops grown after faba bean (Jensen, Peoples and Hauggaard-Nielsen, 2010).

Growing grain legumes such as faba bean does bring with it the risk of post-harvest nitrate leaching and N₂O emissions, meaning that the use of preventive measures, such as catch crops, intercropping, or no-till technologies is desirable (Jensen, Peoples and Hauggaard-Nielsen, 2010). Legumes grown as cover crops can reduce leaching and N₂O emissions compared to bare fallow (Jensen et al, 2012) (Tonitto, David and Drinkwater, 2006). More information on the effects of nitrogen fixing crops on greenhouse gas emissions is given in the literature review on that subject.
A study covering regions of Germany, France, Switzerland and Spain found nitrogen-fixing grain legumes, grown in rotation with cereal crops, could reduce the use of nitrogen fertiliser as its use was not needed on the grain legumes and the residual nitrogen in the soil reduced the requirement for nitrogen fertiliser on the subsequent cereal crop (Nemecek et al, 2008).

Deep rooted nitrogen fixing legumes such as field bean and soybean have been found to loosen compacted soils (Kahnt, Hijazi and Rao, 1986). Field beans decreased the soil bulk density by up to eight per cent, increased total porosity by up to eight percent and air filled pores by up to 19 percent in the top 30 centimetres of soil.

Green fodder and green manure legumes are very suitable for increasing soil organic matter, due to their long growing season, high root biomass, and the practice of leaving the residues in or on the soil, and the crops can be associated with use of minimum or no-till techniques, which reduce soil organic matter loss. Grain legumes such as lentil produce low amounts of dry matter and have little impact on soil organic matter (Underwood and Tucker, 2016). Some further information on legumes as cover crops can be found in the cover crop section of this review.

A review of the potential impacts on biodiversity of EFA choices (Underwood and Tucker, 2016) found that legumes can be attractive to soil macro-invertebrates because of the input of nitrogen rich residues into the soil and because legume roots stimulate soil microbial activity. There is evidence that soil macro-invertebrates are more abundant under forage/green manure legume crops than non-legume crops. For example, a study in Sweden found that alfalfa leys had higher mean population densities and biomass of above ground arthropods, particularly detritivores/microvores, than grass leys (Curry, 1986). A study of boreal grass and legume bioenergy crops found that there were more surface feeding (anecic) earthworms under four-year galega (Galega officinalis) than under a grass crop and bare fallow (Epie, Cass and Stoddard, 2015). Another plot-scale experiment found more earthworms under four-year clover than under grass, though micro-arthropod abundances were similar (Crotty et al, 2015).

However, as the reviewers pointed out, the evidence refers mainly to multi-year forage legume plots, which are expected to be rarely implemented on EFAs. These plots have reduced soil disturbance and an accumulation of litter on the surface, and so provide very different soil conditions to those found under either grain legume crops or leguminous cover crops.

Grain legume crops are subject to weed competition when small and to a range of pests and diseases throughout their growing period. A range of herbicides, fungicides and pesticides are frequently applied (PGRO, 2015). The reviewers point out that these applications may also reduce any benefits that grain legumes have for soil biodiversity. There is, for example, evidence that earthworms may be adversely affected by both fungicides and insecticides and that insecticides can affect soil collembola (Jänsch et al, 2006), though the significance of these effects under normal field conditions is not clear.

The evidence reviewed suggests that the main benefit to soils from growing nitrogen fixing crops is the reduced need for nitrogen fertiliser to sustain arable cropping, though careful husbandry is needed to avoid post-harvest or post-cultivation losses. Multi-year forage legumes have a wider range of benefits for soils than grain legumes.

3.4.11 Creation of "buffer zones" for high nature value areas, Natura 2000 or other biodiversity protection sites, including along hedgerows and water courses

Soil aspects are covered under buffer strips

3.4.12 Borders, in-field strips and patches managed for wildlife or specific fauna (herbaceous border, protection of nests, wildflower strips, local seed mix, unharvested crops)

Soil aspects are covered under buffer strips and fallow.
3.4.13 Production on arable land with no use of fertiliser (mineral fertiliser and manure) and/or plant protection products, and not irrigated, not sown with the same crop two years in a row and on a fixed place;

No direct evidence found for the effect on soils of this particular practice. Using the evidence from the crop rotations and cover crops sections it can be stated that a diverse crop rotation could help to maintain soil quality, especially if leguminous crops formed part of the rotation and cover crops are used. Not keeping the same crop in place for more than one year would limit the opportunities to use fertility-building leys, which are normally left in place for more than one year.

3.4.14 Conversion of arable land into permanent grassland extensively used.

Soil aspects are covered under the permanent grassland measure.
4 LITERATURE REVIEW ON THE EFFECTS ON WATER OF FARMING PRACTICES RELEVANT TO THE PILLAR 1 GREENING MEASURES

4.1 INTRODUCTION

Agriculture is estimated to contribute 55% of the non-point source pollution of surface waters in the EU. Agriculture, as the dominant land use in most parts of the EU, also has a major impact on groundwater recharge. The impacts of agriculture on soil and water are closely connected, and much of the information identified here is also of relevance to impacts on soil. Less than half of the nitrogen and phosphorus in commercially applied fertilizer is taken up by crops. Runoff from agriculture can elevate nitrogen and phosphorus levels in waterways, leading to eutrophication, with negative impacts on water quality, freshwater biodiversity and human health. Dissolved organic nitrogen and mineral nitrogen in soil may both leach, but the former may leach at a greater depth (Bhogal et al, 2000) and is less dynamic (Mengel, Schneider and Kosegarten, 1999; Murphy et al, 2000). Contributing factors may include the excessive application of fertiliser or manure, poorly timed application, autumn ploughing and high livestock stocking density. Nitrogen is generally the main limiting factor for phytoplankton growth in marine systems and phosphorus in freshwater systems.

Leaching processes can also convey pesticides into groundwater and surface water. Some of these pesticides may persist in aquatic systems for many years after being banned, as shown by a study on a Portuguese lake in the Natura 2000 network (Abrantes, Pereira and Gonçalves, 2010). Fertiliser and pesticide use also result in heavy metal inputs to farmland which may end up in surface or ground water. Heavy metals were not covered in this review.

Phosphate leaching from farmland is very closely tied to soil erosion. Dissolved organic carbon concentrations also have a significant effect on water quality. DOC removal may make a large contribution to the cost of water treatment, and incomplete removal can result in the formation of trihalomethanes. These are potential carcinogens and their concentration in drinking water is restricted. Another component of aquatic pollution is suspended solids, which decrease the aesthetic quality of drinking water. Phosphate leaching, DOCs and suspended solids in water are very closely bound to rates of soil erosion, and are therefore primarily covered in the separate soil literature review, and only few examples are cited here.

This literature review focuses on the impacts of greening-relevant agricultural management techniques on the pollution of groundwater and surface water in the EU, specifically:

- Nitrate leaching
- Phosphate leaching
- Pesticide residue leaching
- Groundwater recharge

The scope of the review was limited by the available information of the impacts of greening measures on water quality and quantity. Evaluating the impacts of agriculture on water requires extensive spatial and temporal monitoring processes, as these impacts may be widely distributed through hydrological systems and landscapes, and these systems may take years to respond to changes in land management. As a consequence, many of the findings discussed here are based on smaller-scale experiments which may not reflect longer-term and more spatially distributed impacts. While certain features appear relatively well-studied in terms of their effects on water, such as crop rotation, catch and cover crops and buffer strips, no relevant findings could be identified for stone walls, and very little for terraces, ditches and ponds.

Most of the available literature focused on the role of nitrogen leaching into water courses adjacent to arable land. Few studies addressed impacts of agriculture and greening measures on lakes, for which phosphate is generally a more important pollutant than nitrogen (Christensen and Kørnøv, 2011). There are significant knowledge gaps about the behaviour of other agricultural chemicals with potentially damaging effects on water quality and aquatic life (Borggaard and Gimsing, 2008).
4.2 **Crop Diversification: Greening Measure**

### 4.2.1 Diversification of Crops

The impacts of crop diversification will depend on the relative effects of the initial crop and its (partial) replacement crop. Impacts on water quality can thus be positive or negative.

Introducing a legume as a replacement crop can have positive impacts on water quality by decreasing the amount of fertiliser and pesticides that need to be applied. However, legume crops can lead to nitrate leaching if they are harvested during a period of high rainfall and are not immediately followed by a crop that absorbs the nitrogen added to the soil by the legume (see section on nitrogen-fixing crops for evidence).

The levels and types of pesticides used on different crops varies widely, and also varies significantly between farming systems. For example, glyphosate use intensity was found to differ substantially between the study districts and crops on north Germany arable farms (Andert, Bürger and Gerowitt, 2016). Crop diversification can make a bigger contribution to natural pest control in cases where the wider landscape is structurally simple (Gabriel et al, 2006) than complex.

### 4.2.2 Crop Rotations

Crop rotations have an indirect impact on water quality, as they can affect nitrogen leaching, pesticide residues, erosion, evaporation and irrigation (BIO Intelligence Service, 2010). The quantity of nitrogen fertiliser and pesticides applied can affect the level of runoff into water.

**Nitrogen leaching**

Nitrogen leaching from crop rotations can be reduced by incorporating catch crops, reducing tillage, and cultivating crops with a large autumn root mass. Rotations featuring crops that reduce soil erosion will have positive impacts on water quality. Crops with a high soil cover index offer the most benefits, and rotations can also be chosen to maximise soil cover over the year, as with the use of catch and cover crops. See the section on cover crops below for the evidence of impacts on water quality.

Ulén et al (2005) measured losses of phosphate in 15 crop rotations on a clay soil in southwest Sweden. In crop rotations containing green manures, green fallow or leguminous leys, there was a risk for increased P losses after these crops were ploughed in. The losses increased in the order: cash crops < dairy with grass < dairy with lucerne < monoculture with barley < organic farming with cattle slurry < stockless organic farming with green manure.

**Pesticide residue leaching**

Appropriately chosen crop rotations can control pest and weed levels, and thus reduce pesticide usage. However, the effects of crop rotation may also be negative or neutral, if monocultures and pesticide-dependent rotations are used, depending on whether these pesticides are applied in periods of low or high precipitation and runoff.

### 4.2.3 Winter Soil Cover

A meta-analysis of studies of crop rotations with leguminous or non-leguminous winter cover crops, or no cover crops, compared the resulting yields, N leaching and soil nitrate status after harvest (Tonitto, David and Drinkwater, 2006). The non-leguminous cover crops and bare winter fallow received N fertiliser, while the leguminous ones did not. Nitrate leaching was reduced by an average of 40% in the legume crop systems and by 70% in the non-legume systems. However, post-harvest nitrate status was similar in both systems. See cover crops below for more evidence.

### 4.2.4 Catch Crops

See cover crops below.
4.3 MAINTENANCE OF PERMANENT GRASSLAND GREENING MEASURE

Permanent grassland is often used as pasture or growing feed for livestock such as cattle and sheep. Ploughing and reseeding grassland is known as renovation and may be undertaken to improve productivity.

Water infiltration rates on permanent grassland can be particularly affected by cattle grazing in the winter months. High stocking densities increase runoff and decrease infiltration, and can thus reduce rates of recharge to groundwater resources (Heathwaite, 1990; James and Roulet, 2007)

4.3.1 NO PLOUGHING OF PERMANENT GRASSLAND

Ploughing of permanent grassland results in increased nitrate leaching. Where permanent grassland is ploughed on wetland sites, N losses are particularly high. However, it is worth noting that nitrate leaching from long-term unploughed permanent grassland can be relatively high. A study on grassland sites in the North Wyke model farm in England found that the site with the longest history of permanent grassland had the greatest sediment and nutrient losses, and that careful management was required to avoid the accumulation of surface nutrients and soil compaction (Peukert et al, 2016). The authors explained this result by the fact that this field had (i) the largest nutrient concentrations associated with the surface accumulation of nutrients applied over time and available for mobilization and (ii) the largest mobilization potential of those nutrients to be moved.

4.3.2 RESEEDING PERMANENT GRASSLAND (INCLUDING WITH AND WITHOUT PLOUGHING)

Nitrate leaching

Renovation of grassland may lead to a temporary spike in soil mineral N linked to three causes: ploughing stubble and roots into the ground which are then mineralised, reduced crop uptake of N after ploughing, and enhanced mineralisation after ploughing. Ploughing grassland disrupts soil structure and increases aeration, leading to rapid conversion of soil organic matter to outputs including other forms of N. This higher concentration of mineral N can increase N leaching. On wet and peat grasslands it is likely that ploughing will be preceded by drainage, which will have additional negative impacts on water quality.

A study in Scotland, UK, found a temporary spike in soil mineral N to more than 200 kg ha\(^{-1}\) following the ploughing of one year old clover-rich and grass-dominated swards, previously subject either to cutting or grazing regimes (Davies, Smith and Vinten, 2001). The study concluded that leaving swards ungrazed and unfertilised over winter before ploughing in spring has the potential to reduce nitrate leaching considerably, and that N release intensity is a function of grazing intensity and history prior to ploughing rather than of sward composition. A study in Ireland investigated the effects of ploughing and reseeding permanent grassland in a clay loam soil, finding that renovation increased leaching of organic N to groundwater, with the effect remaining significant for more than a year and decreasing with time (Necpálová, Casey and Humphreys, 2013). Although N mineralization levels were high, the proportions lost to leaching and N\(_2\)O emissions were small, at 4.8 and 0.49% of the N input respectively, likely due to the soil characteristics such as anoxicity. Another study in the Netherlands found that renovation increased leaching in autumn but not in spring (Velthof et al, 2010). A study in Wales (Roberts, Hudson and Roberts, 1989) found that disc harrowing and reseeding upland permanent grasslands increased the loss of nutrients, especially nitrogen, and could thus negatively impact water quality. Reseeding using a minimum cultivation technique reduced these losses, particularly for nitrogen.

Broad-spectrum herbicides may be used to kill grassland vegetation instead of ploughing. There is some evidence to suggest that soil mineral N contents may be even higher without ploughing than with it (Velthof et al, 2010). Other studies found that no-till had mixed effects, which may depend on the specific soil properties and climate (Goss et al., 1990; (Hansen and Djurhuus, 1997).

Phosphate leaching
Ploughing and reseeding permanent grassland can greatly increase the release of particulate phosphorus in high-rainfall areas. However, a study in the UK (Butler and Haygarth, 2007) found that once the sward had developed and was less vulnerable to soil erosion, total phosphorus losses were less than the average for permanent grassland (approximately 0.3 compared to 1 kg P ha\(^{-1}\) year\(^{-1}\)).

### 4.3.3 Ploughing in one area and converting land to PG in another

**Nitrate leaching**

A 4-year study in Lithuania compared nitrate levels in drainage channels and upper groundwater coming from permanent grassland, unfertilized grass leys established in an arable crop rotation and fertilized arable land (Mašauskas et al, 2006). The authors found that nitrate levels in groundwater from permanent grassland were below the 50 mg L\(^{-1}\) maximum level set by the EU Drinking Water Directive. The newly converted land required a year to reach the levels of the permanent grassland. Levels in arable land exceeded the limit on occasion, and at almost all measurements in drainage water. This suggests that ploughing in one area and converting to permanent grassland in another will not balance out in terms of the impact of nitrogen leaching, and will have a net negative effect due to the time taken for newly converted land to approach a lower level of nitrogen leaching, while the ploughed land will have higher leaching rates.

### 4.4 Ecological Focus Areas

#### 4.4.1 Land lying fallow

The effect of fallow land on water quality depends upon the category of fallow, as it may be bare land, land with spontaneous vegetation, or land with green cover. As discussed in the soil literature review, soil erosion and thus the expected runoff will be greater in the case of bare fallow than in arable or covered land (IEEP, 2008). In comparison to arable land, fallow land is not subject to fertiliser or pesticide use, and associated water pollution would be expected to be lower. Intensive grazing degrades soil quality and can increase runoff.

**Nitrate leaching**

A meta-analysis of irrigated systems based on 279 observations and 44 articles found that replacing fallow land with a non-legume crop reduced N leaching by 50% while replacing it with a legume did not affect leaching (Quemada et al, 2013a).

#### 4.4.2 Terraces

Appropriate management of terraces offers benefits in terms of soil structure, and decreases erosion, which results in better water quality, as reviewed in the SoCo report (Louwagie, Gay and Burrell, 2009b). Industrial terracing can have negative impacts on soil structure and increase erosion if the drainage system is not well-designed. Terrace abandonment can result in higher levels of erosion and runoff until pioneer species colonise the site. See the soil literature review for more information.

#### 4.4.3 Landscape features

##### 4.4.3.1 Hedgerows and wooded strips

Hedgerows, like buffer strips, have been shown to act as barriers to surface runoff (Merot, 1999). Their role in intercepting rainfall can reduce water infiltration into soil and groundwater recharge (Le Maitre, Scott and Colvin, 1999).

**Nitrate leaching**
A study on an oak hedgerow in France found that it had no effect on nitrate and salinity levels in deep groundwater, although it appeared to decrease nitrate levels and raise salinity in soil and shallow groundwater (Grimaldi et al, 2012).

4.4.3.2 Isolated trees
No evidence found.

4.4.3.3 Trees in a line
No evidence found.

4.4.3.4 Trees in groups and field copses
A study found that tree shelterbelts can greatly reduce runoff rates (Carroll et al, 2004).

4.4.3.5 Field margins
The evidence here pertains to the function of field margins as buffer strips and is discussed in 1.4.4.

4.4.3.6 Ponds
Passy et al (2012) conducted a modelling study using the Seneque/Riverstrahler biogeochemical model for the Seine, and found that the restoration of ponds at a density of 5% of the corresponding agricultural area would reduce the riverine nitrogen export by up to 25% on an annual basis.

4.4.3.7 Ditches
No evidence found.

4.4.3.8 Traditional stone walls
No evidence found.

4.4.4 BUFFER STRIPS
Buffer strips can reduce leaching of nitrogen and phosphorus into waterways. The JRC Sustainable Agriculture and Conservation (SoCo) report found that buffer strips can reduce the volume of suspended solids, nitrates and phosphates transported by runoff to water bodies by 70-90% (Louwagie, Gay and Burrell, 2009b).

Nitrate leaching
A study in Italy found that a buffer strip of trees, shrubs and grass reduced the quantity of runoff (Borin et al, 2005). The concentration of N in the runoff was increased, but losses were reduced from 17.3 to 4.5 kg ha\(^{-1}\) in terms of mass balance. The authors concluded that the effects of the buffer strip on decreasing eutrophication risk were mainly due to decreasing the quantity of runoff. A subsequent study in Italy found that young buffer strips 4-6m wide with a mix of trees, shrubs and grass reduced total runoff by 33%, N loss by 44% and P by 50% whilst a mature buffer strip reduced NO\(_3\)–N and dissolved phosphorus concentrations by almost 100% (Borin et al, 2010). A study evaluated the nitrogen buffering capacities of two narrow riparian strips (5–8 m) along irrigation ditches located in a flat agricultural watershed of northern Italy with intensive arable farming (Balestrini et al, 2011). The nitrate levels within riparian sites were very low, completely disappearing below the ditches, compared to the high mean concentrations measured at the exit of the crop fields (10–90 mg l\(^{-1}\) N-NO\(_3\)–). The trees on the buffer strips were particularly important because their water uptake affects the subsurface flow pattern and contributes to completely removing the nitrate in the riparian zone.

Novara et al (2013) investigated Sicilian vineyards, finding that catch crops reliably increased nitrate levels in soil, but the levels could be managed through the use of buffer strips. Specifically that a 6m buffer strip reduced the nitrate levels by 42% and a 9m strip reduced it by 46%.
Phosphate leaching
A meta-analysis evaluated the effects of grass buffer strips in reducing P runoff to surface waters and concluded that they are effective (Dorioz et al., 2006). Sedimentation in riparian buffers can result in a P retention rate of up to 128 kg P ha\(^{-1}\) yr\(^{-1}\) and plant uptake may temporarily immobilize up to 15 kg P ha\(^{-1}\) yr\(^{-1}\) (Hoffmann et al., 2009). However, dissolved P is less well retained than particulate P and is often below 0.5 kg P ha\(^{-1}\) yr\(^{-1}\), and several studies indicate the release of significant levels (up to 8 kg P ha\(^{-1}\) yr\(^{-1}\)) (Hoffmann et al., 2009). A study in Italy found that a buffer strip of trees, shrubs and grass lowered total P concentrations and reduced total losses by about 80%, whilst soluble P concentrations were unmodified (Borin et al., 2005). The buffer strip also reduced the concentrations of total suspended solids. A study found that riparian buffers appear to have the greatest potential to improve water quality in relatively small (first-order) streams (Tomer et al., 2009).

An English study placed astroturf mats within a range of buffer strips (mid-field hedges, edge-of-field grass strips and channel wetlands). Mat sediment was recorded, there was a considerable range in sediment deposition over the 18-month sampling period with site-average values (based on all mats at a site) ranging from 0.02 ± 0.06 to 1.15 ± 1.88 g cm\(^{-2}\). The site-average total-P content of the <63 μm fraction of the deposited sediment ranged between 616 and 1938 mg kg\(^{-1}\) (average 890 mg kg\(^{-1}\)) (Owens et al., 2007).

An English study Stutter, Langan and Lumsdon (2009) reported that the establishment of vegetated buffer strips (VBS) in plots in England led to enhanced rates of soil P cycling, increasing soil P solubility and the potential amount leached to watercourses. Soil in VBS, relative to adjacent fields, had increased inorganic P solubility indices, dissolved organic P, phosphatase enzyme activity, microbial diversity, and biomass P.

Pesticide residue leaching
Buffers can contribute to the degradation of pollutants, including pesticides. A study in Italy found that buffer strips 4-6m wide with a mix of trees, shrubs and grass reduced total runoff by 33%, N loss by 44% and P by 50% whilst a mature buffer strip reduced herbicide concentrations by 60% to 90%, depending on the chemical and the time elapsed since application, compared to fields with no buffer strips (Borin et al., 2010). A Danish study concluded that buffer strips had a strong effect on modulating pesticide concentrations in 14 first- and second-order streams (Rasmussen et al., 2011), and the width was important for this. Riparian buffer strips of uniform width may not be adequate in mitigation, due to variations in the flow properties of water courses (reviewed in (Polyakov, Fares and Ryder, 2005)). Such effects can reduce trapping efficiency from 41-99% to 15-43%, and variable widths may be more efficient. Numerous studies have demonstrated the significant (although variable) effectiveness of grass buffer strips at reducing pesticide transfer by surface runoff from agricultural sites to rivers in a variety of cultivation conditions (Lacas et al., 2009). Krutz et al (2005) performed an extensive literature review including studies investigating natural rainfall, simulated rainfall, and simulated run-on experiments. The review found that although results varied according to land, soil and climate type, vegetative buffer strips reliably reduced herbicide run off to adjacent soil in all conditions of their implementation.

Passeport et al (2013) measured the concentration and load transfer of 16 pesticides out of a tile-drained catchment (Bray, France) and their reduction through two buffer zones: an artificial wetland (AW) and a forest buffer (FB). They reported that inlet vs. outlet pesticide load reductions ranged from 45% to 96% (AW) and from −32% to 100% (FB) depending on the pesticide molecule and the hydrological year that ranged from 45% to 96% (AW) and from −32% to 100% (FB) depending on the pesticide molecule and the hydrological year.

Reichenberger et al (2007) conducted a review of 180 studies, and concluded that effectiveness of grass buffer strips in reducing pesticide leaching had been consistently demonstrated, although the levels of reduction varied greatly (mitigated by soil type, landscape, climate etc.). Furthermore the review illustrated that vegetative buffers were more effective at reducing pesticide runoff and erosion inputs into surface waters than riparian buffer strips.
4.4.5 **AGROFORESTRY**

Trees can contribute to reducing N and P leaching through recapture by their roots, at a soil depth greater than that of crops (Allen et al, 2004; Nair et al, 2007). Agroforestry and grass buffers help to reduce non-point source pollution from pastured watersheds. Agroforestry also appears to reduce water content and increase infiltration and storage, according to a study exploring the effects of agroforestry buffer strips (4.5 m wide) on a watershed area previously under a maize/soybean rotation (Anderson et al, 2009).

*Nitrate leaching*

Nitrate yields were lower in forest and pasture watersheds in Spain, compared to grain-sown watersheds (Casalí et al, 2010). The SoCo report found evidence that agroforestry systems reduce leaching (Louwagie, Gay and Burrell, 2009b). Nitrogen leaching could be reduced by up to 28 per cent in areas of high leaching (more than 100 kg N year⁻¹), but this was dependent on tree density. A study in Florida investigated soils under range and pasturelands with and without tree growth in Florida, and concluded that the introduction of silvo-pastoral systems would enhance nutrient retention and reduce nutrient transport to surface water, due to the more extensive rooting zones of the combined tree and forage system (Nair et al, 2007). Research on a silvo-arable agroforestry system in France found that the trees reduced nitrate losses by 36%, due to the increased infiltration of water facilitated by the tree roots compared to crop land without trees (Béduneau and Gabory, 2012b).

4.4.6 **STRIPS OF ELIGIBLE HECTARES ALONG FOREST EDGES**

No evidence found.

4.4.7 **SHORT ROTATION COPPICE**

*Nitrate and phosphate leaching*

A literature review of SRC impacts on water reported positive effects of SRC on groundwater quality in terms of nutrient leaching (Dimitriou et al, 2009). Willow trees grown as short rotation coppice (SRC) in Sweden had significantly lower nitrogen leaching, but higher phosphorus leaching, compared to cereals (Dimitriou, Mola-Yudego and Aronsson, 2012). A study in Germany found that willow and poplar SRCs did not have a positive effect on leaching, and an older willow SRC increased leaching (Schmidt-Walter and Lamersdorf, 2012). The authors also highlight that leaching rates are expected to be higher after the initial planting, due to the ploughing required.

A case study of SRC in a lowland drinking water catchment area near Hannover, Germany found that the nitrate leaching on the fallow ground but also under a 15 year old poplar plantation was significantly reduced (<1 – 2mg NO₃-N L⁻¹) compared to the catchment level (9.5 mg NO₃-N L⁻¹) (Lamersdorf, 2012). Nevertheless, nitrate leaching increased temporarily after site preparation measures for planting (deep ploughing; >15 mg NO₃-N L⁻¹) and was permanently enhanced when a willow plantation was not harvested early enough (10 mg NO₃-N L⁻¹). At 655 mm of annual rainfall, ground water recharge was reduced by 40 % (to 180 mm a⁻¹) through a five year old willow plantation, compared to the fallow ground reference plot (300 mm a⁻¹). However, transpiration was limited by low soil water storage capacities, which in turn led to a moderate impact and transpiration losses.

A UK study (Goodlass et al, 2007) found that the destroying of current crops to plant a SRC caused significant nitrate leaching, as did the final grubbing of the SRC. However the reduction of nitrate leaching the SRC caused over its life span (15-30 years) offset this initial and final leaching, both in general and especially in comparison to a control arable rotation. Langeveld et al (2012) compared an intensive arable crop cultivation to a SRC rotation on 20% of arable land, finding that the SRC significantly reduced groundwater nitrate and surface runoff. However it needs to be noted that SRC’s are mediated by the application of N fertilizers. Balasus et al (2012) found that on both Poplar and Willow SRC the application on N mineral fertilizer caused an average increase of N leaching in the willow and poplar plots of 25 kg N ha⁻¹ year⁻¹ and 40 kg N ha⁻¹ year⁻¹, respectively. The application of fertilizer had no impact on yield, so it is recommended that N fertilizer is not used on SRC plantations.
Groundwater recharge

Groundwater recharge rates were not greatly impacted by SRCs in a German study and the authors recommend the use of short rotation cycles to maximise resprouting and improve recharge rates (Schmidt-Walter and Lamersdorf, 2012). Another case study of SRC in a lowland drinking water catchment area in Germany found that ground water recharge was reduced by 40% (to 180 mm a-) through a five year old willow plantation, compared to the fallow ground reference plot (300 mm a-) (Lamersdorf, 2012). However, transpiration was limited by low soil water storage capacities, which in turn led to a moderate impact and transpiration losses. A multi-criteria assessment study of poplar SRCs in Germany found that small SRCs had a lower impact on groundwater recharge rates than larger SRCs and were preferred by farmers (Busch, 2012).

4.4.8 Afforested Areas

Nitrate leaching

Afforestation of agricultural land can be used as a strategy to improve water quality. A study evaluated the effect of afforestation of former arable land in Denmark on nitrate leaching, based on three afforestation chronosequences (Hansen et al, 2007). The results suggest that afforestation of former arable land initially leads to lower nitrate leaching than that occurring under the former agricultural land use, and largely below the standard of 50 mg NO–3 L–1 for groundwater to be utilized as drinking water. Nitrate concentrations became almost negligible in forest stands of 5–20 years of age. However, after canopy closure (>20 years) nitrate concentrations below the root zone and nitrate leaching tended to increase, attributed to increased N deposition with increasing canopy development and decreased N demand because of the reduced rate of tree growth.

Groundwater recharge

Afforestation can have a negative impact on water-scarce areas, or in areas such as reservoir catchments where a high runoff yield is desirable, because of the increased evaporation rates and infiltration (Hell, Muys and Hansen, 2007). A review on afforestation of agricultural land found that in general its impacts on water quantity are to reduce average streamflow and low flows, reduce groundwater recharge, which can affect downstream water security to an extent, but at the same time the afforestation may enhance local to regional rainfall (van Dijk and Keenan, 2007). Potential impacts on water quality are to reduce the volume of sediment, nutrients and salt, but not necessarily pollutants, and it can restore soil function in degraded catchments. A study of four afforestation sites in Denmark simulated water recharge from the hydrological measurements and found that it decreased with increased stand age within 30–40 years of afforestation, mainly due to increased interception evaporation with tree age (Rosenqvist et al, 2010).

A study simulated the effect of afforesting arable farmland around the second largest aquifier in the UK. They found that conversion of 80ha and 240ha of agricultural land to woodland could potentially restrict nitrogen losses to the point that the groundwater would satisfy the EU groundwater nitrate limit (50mg L-1). Furthermore that if the entire target zone was converted then nitrate concentration would decrease by 35% by 2025 (Zhang and Hiscock, 2011).

4.4.9 Catch Crops / Green Cover

Nitrate and phosphate leaching

There is substantial evidence that catch and cover crops reduce nitrate and phosphate leaching from arable crops (see Box 4-1). Several studies have shown that catch/cover crop seed mixes that combine one or more legume species with a grass or cereal species have reduced nitrate leaching and a more stable N accumulation in soil biomass than a legume-only catch/cover crop, whilst increasing yield and nitrogen content of the subsequent crop more than a non-legume catch/cover crop.
Evidence of the impact of catch crops and green cover on nitrate and phosphate leaching

A meta-analysis of 35 studies conducted in the Nordic countries (Denmark, Finland, Norway and Sweden) found that non-legume catch crops reduced N leaching by 50% on average, while legumes did not reduce leaching risk (Valkama et al, 2015). A study in Ireland found that a mustard winter cover crop significantly reduced nitrate losses for both ploughed and reduced cultivation treatments (Hooker et al, 2008). It further suggests that such cover crops are particularly important in reducing nitrate loss in temperate regions with mild winters, where winter N mineralisation can be high. It is generally accepted that catch crops grown outside the main season in Nordic climates can effectively reduce nitrate and ammonium losses (Doltra and Olesen, 2013). A study in Denmark found that a ryegrass catch crop for barley reduced leaching in the short term (Thomsen, 2005). In a long-term study of three sites in Northern France comparing the effects of catch crops, no-till and fertilizer reduction below recommended levels on parameters including nitrogen leaching, it was found that catch crops had the greatest impact on reducing leaching (Constantin et al, 2010). The main crops were winter wheat, spring barley, spring pea, silage maize and sugar beet, and the catch crops were white mustard, Italian ryegrass, and radish or winter cereal. The authors found that establishing catch crops every year in low-drainage systems sufficed to reach EU water quality standards. A modelling study (Taylor, He and Hiscock, 2016) based on high-frequency water monitoring in a river catchment in England found that a red clover cover crop was the most effective of the agricultural measures investigated and relevant here for reducing nitrate losses (19.6%). This was lower than an earlier finding of 63% (Ullrich and Volk, 2009) relative to a solely conservation tillage scenario. This model benefited from adequate data to inform its predictions. The most effective mitigation option to reduce both nitrogen and phosphorus losses was found to be a combination of cover crops and buffer strips. A study comparing the effects of 17 different catch crops with spring barley in a no-tillage system found that undersown ryegrass was effective in reducing N leaching, but strongly affected barley yield, whilst white and red clover did not affect yield as much, and did not increase leaching (Känkänen and Eriksson, 2007). Askegaard, Olesen and Kristensen (2005) tested three sites in Denmark, finding that catch crops reduced nitrogen leaching by an average of 30-38%, reducing concentrations in water to drinking water standard. (Laurent and Ruelleand, 2011) conducted 9-year simulations in France, which showed as a result of catch crops nitrate flow reduced by 11%, and by 12% following catch crops under a no till system. A Spanish study (Gabriel, Muñoz-Carpena and Quemada, 2012) reported that over a 3.5 year period cumulative nitrate leaching in fallow, vetch, and barley systems was 346, 245, and 129 kg N-NO$_3^-$ ha$^{-1}$, respectively; all of which occurred by more than 77% during the intercrop period. This study also found that vetch was less efficient than barley at controlling leaching, but enhanced soil N retention. Quemada et al (2013b) conducted a meta-analysis of 44 studies, finding that replacing a fallow with a non-legume cover crop reduced nitrate leaching by 50% while using a legume cover crop did not have any effect on leaching.

Mixed Catch Crops

A meta-analysis of Nordic studies found that mixed grass and legume catch crops increased both grain yield and grain N content of subsequent spring cereals by 6%, as opposed to non-legume single species catch crops which reduced grain yield by 3% with no changes in grain N content (Valkama et al, 2015). A winter cover crop barley–vetch mixture was shown to have a more stable N accumulation in soil biomass and reduced NO$_3^-$ N leaching risk compared to single species barley or vetch winter covers in Mediterranean cropping conditions (Tosti et al, 2014). A Swedish study (Bergkvist et al, 2011) investigated spring under-sown clover cover crops with or without perennial ryegrass in the seed mixture, used in a winter wheat and barley system. They found that adding ryegrass to clover mixtures reduced the amount of N leached compared to clover only mixes, but still maintained the nitrogen residue of a clover cover crop.

Pesticide residue leaching

There is considered to be a research gap in investigating the roles of catch and cover crops in mitigating the contamination of water with pesticides, faecal indicator organisms and pathogens (Randall et al, 2015).

4.4.10 NITROGEN-FIXING CROPS

Nitrogen-fixing crops can offer indirect benefits through reducing the amount of fertiliser that needs to be applied, and thus reducing leaching into water bodies (see Box 4-2). However, this may be true only in the growing season, and in the following season their nitrogen-rich residue can lead to leaching (Bues et al, 2013). This effect could be reduced by the use of a catch crop and the early sowing of winter grain legumes. It also appears that the type of legume has strong effects on the level of leaching (Pappa et al, 2011). The authors of (Nemecek et al, 2008) also investigated the effect on ecotoxicity potential of introducing grain legumes into rotations, and found mixed results.
Literature reviews on the effects of farming practices associated with the CAP greening measures on climate and the environment

Box 4-2 Evidence of the impacts of nitrogen-fixing crops on nitrate leaching
A review concluded that introducing grain legumes (peas and soya beans) into crop rotations led to increased N leaching, despite the reduction in N fertilizer applied (Nemecek et al, 2008).
A study found that grass-clover plots leached less N after ploughing than grass-only plots (Davies, Smith and Vinten, 2001).
A modelling study found that increasing the number of grain legumes (i.e. peas, beans) in crop rotations led to greater N leaching, and this was reduced by the use of cover crops (Plaza-Bonilla et al, 2015); fertilizer was not added to grain legumes.
In a study comparing the effects of different legume intercrops for barley, clover had the highest nitrate rate leaching rate, while the rate with a pea intercrop was affected by the pea variety (Pappa et al, 2011).

Pesticide leaching
Grain legumes are susceptible to pest damage and require the application of high levels of insecticides in comparison to cereals (Underwood and Tucker, 2016). Legume forages require lower pesticide levels but a pre-sowing herbicide may be applied (Williams et al, 2014).

4.4.11 Creation of "buffer zones" for high nature value areas, Natura 2000 or other biodiversity protection sites, including along hedgerows and water courses
Buffer zones around small nature protection sites are crucial to protect the biodiversity associated with freshwater habitats and low nutrient habitats such as bogs. A study in Denmark found that nature protection areas vulnerable to ammonia eutrophication can be positively affected by introducing buffer zones around them (Schou et al, 2006).

4.4.12 Borders, in-field strips and patches managed for wildlife or specific fauna (herbaceous border, protection of nests, wildflower strips, local seed mix, unharvested crops)
Margins can help to reduce pesticide use by providing a measure of biological control. They can also decrease runoff and influence the movement of nutrients and water within the system. It has been shown that a 3m-wide buffer strip reduced pesticide drift into an adjacent ditch by 95% and a 6m-wide strip removed all measurable drift (de Snoo and de Wit, 1998), (De Snoo, 1999), reviewed in (Marshall and Moonen, 2002).

4.4.13 Production on arable land with no use of fertiliser (mineral fertiliser and manure) and/or plant protection products, and not irrigated, not sown with the same crop two years in a row and on a fixed place;
Not reviewed.

4.4.14 Conversion of arable land into permanent grassland extensively used
Not reviewed.
5 LITERATURE REVIEW ON THE EFFECTS ON GHG EMISSIONS OF FARMING PRACTICES RELEVANT TO THE PILLAR 1 GREENING MEASURES

5.1 INTRODUCTION

Agricultural land management has an impact on GHG emissions, both as a source of GHG emissions as well as a sink, through carbon storage and sequestration. The agriculture sector remains a significant emitter of methane (CH\textsubscript{4}) and nitrous oxide (N\textsubscript{2}O) emissions (10.2% for the EU-28 – 2014 figures) and in some countries these constitute a considerable proportion of total greenhouse gas emissions (ranging from around three per cent in Malta to 32 per cent in Ireland). If emissions and removals from soils are taken into account agriculture accounts for 13 per cent of total emissions for the EU-28 (2014 figures). In terms of CO\textsubscript{2} emissions, cropland management overall is a net source of emissions (although not in all countries) and is predicted to remain a source in the future. Grazing land management overall for the EU-28 is a net sink (although in the past it has been a source of emissions) and is predicted to remain a sink in the future.

Nitrous oxide (N\textsubscript{2}O) emissions are caused largely by application of nitrogen (N) fertilizers and livestock manures to land, and methane emissions (CH\textsubscript{4}) result from enteric fermentation and manures as well as rice cultivation. The literature reviewed identified a range of agricultural practices that can promote the reduction in emissions or increase in removals of GHG emissions from the atmosphere and provide estimates of their mitigation potential in different situations. However, these values can vary significantly in different parts of the EU, depending on the soil type, climate (arid, wet) and existing land use.

This literature review attempts to summarise the evidence for the benefits of the full range of land management practices that have the potential to be supported via the three standard greening measures and equivalent practices on GHG emission mitigation within the European Union. It focuses on both the role of land management practices on reducing GHG emission sources and enhancing carbon sinks.

There have been a number of studies in recent years that have carried out systematic reviews of the literature on the GHG mitigation potential of a range of agricultural management practices, many of which are those included under the greening measures (Martineau et al, 2016); (Frelih-Larsen et al, 2014); (Schils et al, 2008). These sources have been the primary source of information for this literature review. These have been supplemented by following up key references quoted in these meta-reviews of the literature and using IEEP’s own reference database and Google Scholar to supplement this information. Where information is relevant for soils and GHG emissions reductions, the information is covered in both reviews.

5.2 CROP DIVERSIFICATION GREENING MEASURE

5.2.1 DIVERSIFICATION OF CROPS

The main literature sourced focused predominantly on crop rotations rather than crop diversity (see below). However, many of the findings relating to crop rotations are also relevant here. In particular, the climate mitigation effects of diversifying crops will depend on the type of crops cultivated and the extent to which these decrease the amount of fertiliser that needs to be applied. Effects on reducing N\textsubscript{2}O emissions are most likely if legume crops or fallow are introduced into the system. The effects of these for GHG emission reductions are covered under the EFA section below.

5.2.2 CROP ROTATIONS

The key elements of crop rotations that have a bearing on GHG emissions are:

- The inclusion of catch crops or green manures in the rotation leading to less bare land and more green cover over winter;
- The type of crop species cultivated;
The addition of legumes / N fixing crops to the rotation or undersowing with these crops (BIO Intelligence Service, 2010)

Frelih-Larsen et al (2014) consider the mitigation potential of “incorporating 1–3 years of a perennial crop (often alfalfa or grass hay) into annual crop rotations”. The five main ways that perennial crops can mitigate GHG emissions are given as follows:

1. Sequestering C to soil.
2. Reducing direct emissions from N fertilizers.
3. Reducing the CO₂ emissions from fertilizer manufacture.
4. Reduction in N leaching.
5. Reduction in the amount of N that needs to be applied to the following crop.

The impacts of rotations on N2O emissions depend largely on the type of crops included in the rotation and the soil coverage achieved. The presence of legumes and crops that require lower doses or no N fertiliser will be more beneficial as will rotations where the soil is covered throughout the year, but especially in the winter to limit N leaching.

Estimates of mitigation potential range from 0 to 1.2 t CO₂ ha⁻¹ yr⁻¹, for carbon sequestration in soil, and a saving of 0.7 t CO₂e ha⁻¹ yr⁻¹ from other sources such as N fertilizers, field operations, and N₂O emissions from soil (Frelih-Larsen et al, 2014). However, it is not made clear whether the sequestration estimate takes account of cultivation on return to annual crops, and whether the area used in the calculations includes the whole rotation or just the area of perennial crops. Franzluebbers, Sawchik and Taboada (2014) provide evidence from North and South America that the inclusion of perennial forage crops in a rotation, compared with a rotation that did not include a perennial crop, has a long-term benefit of increased SOC over several rotations, i.e. the benefit was maintained beyond the duration of the perennial crop.

McDaniel, Tiemann and Granddy (2014) conducted a meta-analysis of 122 studies, and concluded that the adding of one or more crops into rotation of a monoculture system increased total soil C by 3.6% and total N by 5.3%. When rotations included a cover crop the total C increased by 8.5%, and total N 12.8%. In addition to this they found that rotations substantially increased the soil microbial biomass pools of C by (20.7%), and N by (26.1%). The GHG emission effects of crop rotations is highly sensitive to the share of land used for perennial crops, and the type of perennial crops used. Carbon sequestration is also influenced by the quantity of N fertilizer used (Franzluebbers, Sawchik and Taboada, 2014); more N fertilizer leads to greater biomass production. Plant density (sowing rate) and biomass production rate of the crop are other factors that have an influence on the overall effect of the action - denser crop stands are likely to provide greater soil carbon sequestration, as are those with high biomass production, although a large proportion of this will be lost once the crop is harvested, depending on whether crop residues are ploughed back into the soil or not.

5.2.3 WINTER SOIL COVER

See EFA section on cover crops

5.2.4 CATCH CROPS

See EFA section on catch crops

5.3 MAINTENANCE OF PERMANENT GRASSLAND GREENING MEASURE

This section covers both the maintenance of grassland under the ratio rules and the protection of ESPG inside and outside N2K areas
5.3.1 NO PLOUGHING OF PERMANENT GRASSLAND

A number of practices have been highlighted in the literature that could contribute to increasing the carbon content of soils under grassland (summarised in O’Mara (2012) – see below. The increase in carbon sequestration by improved grassland management will vary considerably depending upon:

- the management practice adopted;
- previous management;
- soil type; and
- climate.

Reported estimates range from 0 to over 2.0 t C/ha/year (Buckingham et al, 2014). The uncertainties concerning the estimated values of C storage or release after a change in grassland management are still very high (estimated at 25 g C/m² per year).

Grazing intensity. The intensity and timing of grazing (and livestock species) can influence the removal, growth, carbon allocation, and flora of grasslands, thereby affecting the amount of C accrual in soils. The literature shows that the effects of grazing intensity are not consistent. Both under- and over- grazing can reduce carbon sequestration or lead to carbon losses from soils (Rice and Owensby, 2001); (Liebig et al, 2005). The CLIMSOIL report (Schils et al, 2008) attributed this variation to the many types of grazing practices employed and the diversity of plant species, soils, and climates involved. Follett and Schuman (2005) reviewed grazing land contributions to C sequestration worldwide using 19 regions. A positive relationship was found, on average, between the C sequestration rate and the animal stocking density. However recent studies have shown that low intensity grazing increases carbon sequestration, while reducing GHG emissions from soil (N2O) and from livestock (CH4) (Soussana, Tallec and Blanfort, 2010); (Follett and Schuman, 2005) found that across sites, net C sequestration declined with the degree of herbage utilisation by herbivores through grazing as well as cutting, indicating that grassland C sequestration per unit area is favoured by extensive management provided that nutrients are not limiting (Allard et al, 2007; Klumpp, Soussana and Falcimagne, 2007).

Increased productivity through enhanced nutrient management and irrigation. Improving the productivity of pastures through practices such as fertilization and irrigation can improve carbon storage in pastures. However, some of these gains can be offset by N2O emissions from N fertilizers and the energy used in irrigation. Comparisons between management systems have shown that intensively managed grasslands can sequester over 2 t C ha⁻¹ year⁻¹ more than extensive systems (Ammann et al, 2007). This is due to the increased volume of plant biomass produced, both above and below ground. In mineral soils, fertilization of grassland is generally considered to enhance C storage due to enhanced productivity (O’Mara, 2012). A positive correlation between C sequestration and N fertilization has been observed in managed grasslands (Jones, Donnelly and Albanito, 2006). A further meta-analysis of 115 studies in pastures and other grazing lands worldwide (Conant, Paustian and Eliot, 2001), reported in Soussana, Tallec and Blanfort (2010) also indicated that soil C levels increased with improved management (primarily fertilization, grazing management and improved grass species) in 74% of the studies considered. Irrigating grasslands can also promote soil C gains (O’Mara, 2012). Cultivating to re-seed pastures will release some of the C sequestered. However, as highlighted above in relation to increasing grazing intensity, increased fertilisation of grasslands will also lead to increased N2O emissions.

By contrast, the intensification of nutrient-poor grasslands on organic soils may lead to large C losses, and the conversion of permanent grasslands to leys of medium duration will also lead to the release of soil C. One recent study (Ward et al, 2016) has found that, when soil carbon in the top one metre of soil was measured, Soil carbon stocks were higher in intermediate managed permanent grassland than in either extensively managed or intensively managed permanent grassland

Enhancing grass species diversity. In particular, introducing new deep-rooted grasses with greater productivity can increase soil carbon, particularly on low-productivity pastures (Tilman, Hill and Lehman, 2006).
5.3.2 **RESEEDING PERMANENT GRASSLAND (INCLUDING WITH AND WITHOUT PLOUGHING)**

As highlighted in the soils literature review, studies in Ireland (Necpálová, Casey and Humphreys, 2013), northern Spain (Estavillo et al, 2002) and the Netherlands (Vellinga, van den Pol-van Dasselar and Kuikman, 2004) all found that the ploughing and reseeding of permanent grassland led to significant releases of N\textsubscript{2}O into the atmosphere. In Ireland, the study focused on a dairy farm and identified a loss of 3.1 tonnes per hectare of total nitrogen and 13.4 kilogrammes of soluble nitrogen in the 1.7 years after ploughing. In Spain, the study focussed on cattle grazed permanent pasture and found that ploughing promoted soil organic nitrogen mineralisation, which led to an increase in the rates of N\textsubscript{2}O production, N\textsubscript{2}O production from nitrification, N\textsubscript{2}O production from denitrification and total denitrification. These effects were greater when fertiliser was also applied. In the case of the Netherlands ploughing permanent grassland for conversion to ley rotations and for grassland renovation was responsible for considerably increased N\textsubscript{2}O and CO\textsubscript{2} emissions:

- permanent grassland to arable: 250 tonnes of CO\textsubscript{2} equivalent per hectare;
- permanent grassland to a ley/arable rotation: 150 tonnes of CO\textsubscript{2} equivalent per hectare.

Although soil carbon and nitrogen decreased sharply during the brief fallow periods associated with grassland renovation, the emissions from grassland renovation were much less than from conversion to arable. It also demonstrated that emissions of N\textsubscript{2}O could be further reduced by undertaking renovation in the spring.

A study (Tilman, Hill and Lehman, 2006) has shown that enhancing species diversity and, in particular, introducing new deep-rooted grasses with greater productivity into the species mix can increase soil carbon, particularly on low-productivity pastures.

5.3.3 **PLOUGHING IN ONE AREA AND CONVERTING LAND TO PG IN ANOTHER**

The previous section highlighted that ploughing (and re-seeding) permanent grassland increases GHG emissions (CO\textsubscript{2} and N\textsubscript{2}O). The conversion of arable land to grassland has the potential to sequester carbon in the soil. As highlighted above, the CLIMSOIL project (Schils et al, 2008) concluded that soils under grass generally accumulate C, although with a large uncertainty (see previous section). The exact amounts of C sequestered are very difficult to predict as the process depends on soil type, climate and grassland management.

The amount of C that can be sequestered depends greatly on both previous land use, including intensity of N application, soil type (particularly soil clay content), climate and subsequent grassland management. In general, soils previously in long-term tillage, with a small SOC content will be able to sequester more C. Lugato et al., (2014), found median annual rates of sequestration of c. 0.6 t/ha C up to 2020 (between 0.4 and 0.8 t C ha/ yr). The capacity to sequester C increases with increasing clay content - recent work from Scotland reports 31.1 (+/-0.3) g C m\textsuperscript{-2} y\textsuperscript{-1} (Rees, year?). Subsequent grassland management, e.g. fertilizer inputs and intensity of grazing will also influence the C sequestration potential.

However, an equilibrium will be reached after which there will be no further increase in C storage. In terms of the length of time it takes for grasslands to reach a carbon equilibrium, O'Mara (2012) reported the time-scale to range from 30 to 40 years. Other studies have shown that grasslands have a large potential to store additional carbon and may continue to act as a carbon sink for longer periods of time (Poeplau et al, 2011).

5.4 **ECOLOGICAL FOCUS AREAS**

5.4.1 **LAND LYING FALLOW**

Studies on the contribution of fallow land to reducing GHG emissions are relatively few and the effects are mixed. Although some carbon sequestration may take place, the extent to which this happens depends on the length of time the land is put to fallow as well as the vegetative cover on the fallow land. Often cultivated land fallow areas contain negligible plant biomass which limit their carbon sequestration and storage potential. This is highlighted by the 60 year experiment at
Rothamsted in the UK which showed that carbon concentrations in fallow plots were lower than in either the continuous grassland (82% less) or continuous arable (70% less) plots (Wu et al, 2010). As highlighted in the soils literature review, an IEEP review on the effects of set-aside (IEEP, 2008) showed that green cover on set-aside increased soil organic matter and improved soil structure. A study in east Shropshire (Fullen, 1997) looking at soil erosion on sandy soils showed that soil organic matter content declined when the soil was bare, but increased by an average of 0.39 per cent after two years of grass covered fallow and by an average of 0.78 per cent after four years. A modelling exercise looking at the potential impact of grass fallow in south east England and Sweden (Sahrbacher et al, 2015) predicted that planting miscanthus (Miscanthus giganteus) for bioenergy on fallow could result in a faster accumulation of soil organic matter (1.5 per cent per year rather than 0.5 per cent per year) than would be obtained from a normal grass fallow. A study by Gan et al (2012) found that a continuous wheat systems (tested on Canadian prairies) had the highest annual gain in organic carbon, gaining 38%, 55%, and 127% more than those gained in several fallow (F) - wheat (W) rotations (FWWWWW, FWW, and FW systems, respectively). They concluded that the SOC gain more than offset the greenhouse gas (GHG) emissions that occurred during wheat production, leading to negative emission values at an annual value of $-742 \text{ kg CO}_2 \text{ eq ha}^{-1}$.

In terms of N2O emissions, fallow areas are subject to loss of residual N from the previous cropping period, leading to an increased risk of N2O emissions. However, no fertilisers will be used on the area for a period of time which mean lower N2O than would be the case if the area had been continuously cropped. Finally the reversion of fallow to cropping would then lead to a release of CO2 emissions and lead to an increase in N2O emissions through the ploughing in crop residues.

5.4.2 TERRACES

No particular benefit for GHG emission reductions was identified in the literature, beyond the effects of maintaining soil organic matter through avoiding or reducing erosion risk (see soils literature review).

5.4.3 LANDSCAPE FEATURES

5.4.3.1 Hedgerows and wooded strips

Frelih-Larsen et al., (2014) identified from the literature that hedgerows have a moderate effect in the avoidance of carbon loss due to reduced erosion and the increase in carbon stored in arable fields (Bhogal et al, 2009; Posthumus et al, 2015). Higher hedgerows have the potential to sequester a greater quantity of C within plant biomass (Warner, Tzilivakis and Lewis, 2008). Additional C can be sequestered by ‘gapping up’ hedgerows as well as the planting of new or restoration of hedgerows. This is confirmed by a study of the ‘bocage’ landscape in France (Walter et al, 2003) which calculated that a total of between 13 and 38 per cent of the total soil carbon in the bocage landscape was associated with hedgerows (see soil literature review). Folliain et al (2007) conducted field surveys in France, concluding that hedgerows could act as carbon sinks, but the amount of soil organic carbon greatly varied: median value of 13.3 kg C m $-2$, and an inter-quartile lying between 10.5 and 18.5 kg C m $-$. The study highlighted that SOC storage could be locally important in the vicinity of hedges.

5.4.3.2 Isolated trees

The main effect from isolated trees is in sequestering and storing carbon. However, trees only sequester carbon up to a certain equilibrium.

5.4.3.3 Trees in a line

Frelih-Larsen et al. (2014) identified from the literature that shelterbelts have an impact of +14 kg C/ha/yr in relation to the avoidance of carbon loss due to reduced erosion and the increase in carbon stored in arable fields (Bhogal et al, 2009; Posthumus et al, 2015).
5.4.3.4 Trees in groups and field copses
See isolated trees above

5.4.3.5 Field margins
See section on buffer strips

5.4.3.6 Ponds
No particular benefits for GHG emission reductions found in the literature. However, Tournebize et al (2015) tested the use of a manmade pond-reservoir system in France over eight years, finding an average nitrate reduction in soils of 18%, indicating the potential for such combination systems for filtering nitrate from agricultural soils.

5.4.3.7 Ditches
No particular benefit for GHG emission reductions found in the literature

5.4.3.8 Traditional stone walls
No particular benefit for GHG emission reductions found in the literature

5.4.4 Buffer strips
Buffer strips can locally enhance soil carbon storage if properly vegetated and managed. For example Borin et al, (2010) reviewed data from research programmes conducted over the last decade in Veneto Region (North-East Italy) and showed that the different buffer strips monitored stored up to 80 t ha−1 year−1, although these higher rates were seen in those strips which included vegetation of a more woody nature.

Although riparian buffer strips play an important role in reducing nitrogen inputs to surface waters, they may contribute to increasing greenhouse gas emissions by emitting N2O. Most of the literature on buffer strips relates to their function in reducing the leaching of pollutants such as nitrogen and phosphorous into water courses. However, the process of denitrification is also considered as a major source of N2O. A study in the Netherlands (Hefting, Bobbink and de Caluwe, 2003) found that in heavily nitrate loaded grass buffer strips N2O emissions were between 2–4 kg N ha−1 yr−1. They concluded that “N transformation by nitrate-loaded buffer zones results in a significant increase of greenhouse gas emissions”. Hinshaw and Dahlgren (2016) carried out measurements in and adjacent to buffer strips beside agricultural land along the San Joaquin River. N2O emission were higher from the buffer strips than the land adjacent to them, suggesting that “the riparian zones within the agriculturally impacted San Joaquin River were a significant source of N2O when elevated NO3− was present”.

5.4.5 Agroforestry
Agroforestry is the practice of integrating trees and shrubs with the production of food crops or livestock. Agroforestry is known to have an important role in carbon sequestration (Aertsens, De Nocker and Gobin, 2013; Baah-Acheamfour et al, 2014; Oelbermann, Paul Voroney and Gordon, 2004). A meta-analysis (Branca et al, 2013) reviewed 160 global studies, finding that agroforestry significantly increases aboveground C sequestration. These systems are able to store more C than conventional arable systems (Baah-Acheamfour et al, 2014). Freilih-Larsen et al., (2014) reported that agroforestry sequesters 138 kg carbon per hectare per year. However, the type of agroforestry systems and their capacity to sequester C vary globally (Oelbermann, Paul Voroney and Gordon, 2004). Experiments in Vézénobres (France, Mediterranean climate, sandy loam soil) indicated that poplars (140 trees/ha) of 13 years old have on average sequestered 540 kg C/tree in the trunk and 60 kg C/tree in the root system (Aertsens, De Nocker and Gobin, 2013).
However, whilst the agroforestry sector acts as a sink for CO₂, it may also be a source of N₂O and CH₄ emissions (Povellato, Bosello and Giupponi, 2007). A study of afforestation on arable soils (Morris et al, 2007) found that soil N increases were closely related to C accumulation and were two times greater than could be explained by system N inputs from atmospheric deposition and natural sources.

5.4.6 STRIPS OF ELIGIBLE HECTARES ALONG FOREST EDGES

No specific literature was found relating to this management practice. If these strips are sown to grass, these could have similar effects to permanent grassland if sown to grass and maintained over multiple years.

5.4.7 SHORT ROTATION COPPICE

As also highlighted in the soil literature review in northern Europe short rotation coppice (SRC) can enhance levels of soil organic carbon when established on formerly arable soils. One study in southern Germany (Díaz-Pinés et al, 2016) found that, under hybrid poplar, soil organic carbon increased by 400 kilogrammes per hectare over five years. Another study from north east Germany (Kahle et al, 2007) found an increase of 4 grams of carbon per kilogramme of soil after 12 years. A study of 14 long established SRC plots in southern Sweden (Dimitriou et al, 2012) found that organic carbon concentrations in the soils were significantly higher than in adjacent arable fields. Organic carbon in the topsoil was 9 per cent higher and in the subsoil it was 27 per cent higher. Lasch et al (2010) conducted a modelling study on the impacts of aspen SRC plantations in Germany, under both current climate and climate change, projections. They found that the mean soil carbon sequestration was 0.81 t C ha⁻¹ a⁻¹ under current climate, and 0.93 t C ha⁻¹ a⁻¹ under the assumption of climate change induced climate. Pellegrino et al (2011) assessed the impact of a 10-year SRC poplar stand in Italy. They reported a significant increase in soil organic carbon and microbial biomass carbon as a result, with triennial coppicing proving to have the most benefit (comparatively to annual and biannual). A modelling study found that both willow and miscanthus sequestered carbon at a rate directly proportional to the rate of carbon input, also that there SRCs had a higher sequestration potential than the restoration of natural woodland. Finally the study stated that the miscanthus crop showed the highest sequestration potential (Matthews and Grogan, 2001).

The study in southern Germany (Díaz-Pinés et al, 2016) also looked at rates on N₂O emissions and nitrate leaching from hybrid poplar SRC. It found that rates of both N₂O emissions and nitrate leaching declined as the coppice became established. There was a brief increase in N₂O emissions after coppicing, but no significant increase in gross nitrification. Overall losses of nitrogen from soil decreased by 80 percent if no fertiliser was added. Nitrogen losses increased if 40 to 50 kilogrammes of nitrogen fertiliser per hectare were added, but were still 40 per cent less than those from adjacent arable land. The authors concluded that such additions of nitrogen were in any case not necessary, at least in the first cycle of coppice growth. Vande Walle et al (2007) investigated SRC systems of birch, maple, poplar and willow over a four year period in Belgium, in contrast to most other studies, no weed control, fertilisation or irrigation was applied in this experiment. They found that rates varied, but the use of such rotations significantly increased soil carbon and nitrogen.

Several studies have also identified the effect of SRCs on GHG emissions. Zenone et al (2016) for example surveyed a SRC in Belgium of Poplars, (during the first two rotations over a four year period). The overall GHG balance was an emission of 1.90 (±1.37) Mg CO₂eq ha⁻¹; this indicated that soil trace gas emissions offset the CO₂ uptake by the plantation. CH₄ and N₂O contributed almost equally to offset the CO₂ uptake of −5.28 (±0.67) Mg CO₂eq ha⁻¹ with an overall emission of 3.56 (±0.35) Mg CO₂eq ha⁻¹ of N₂O and of 3.53 (±0.85) Mg CO₂eq ha⁻¹ of CH₄. N₂O emissions mostly occurred during one single peak a few months after the site was converted to SRC; this peak comprised 44% of the total N₂O loss during the two rotations. (Sabbatini et al, 2015) found that on a site in Italy a SRC plantation of hybrid poplar converted from cropland acted as a net sink of GHG (−2202±792 g CO₂eq m⁻²), in comparison to non-converted crop land, which was almost neutral (156±264 g CO₂eq m⁻²). Zona et al (2013) studied a SRC poplar plantation that was on converted farmland, during the first six months after the establishment of the plantation (June–December 2010) the study observed substantial CO₂, CH₄, and N₂O emissions (a total of 5.36 ± 0.52 Mg CO₂eq ha⁻¹ in terms of CO₂ equivalents). However, in the second year (2011) there
was substantial net CO2 uptake \((-3.51 \pm 0.56 \text{ Mg CO}_2\text{eq ha}^{-1}\)). Over the entire measurement period, the sum of the CH4 and N2O losses was much higher \((3.51 \pm 0.52 \text{ Mg CO}_2\text{eq ha}^{-1})\) than the net CO2 uptake \((-0.76 \pm 0.58 \text{ Mg CO}_2\text{eq ha}^{-1})\), indicating that the effectiveness of crops as SRC's is highly varied and subject to environmental conditions (water was a strong determinant of GHG emission/extraction in the study).

However one study found that overall there was no evidence that the conversion of cropland to SRC had a general effect on soil organic carbon. For example, the SOC density was significantly higher under SRC \((21 \pm 2 \text{ kg C m}^{-3})\) compared to cropland \((17 \pm 2 \text{ kg C m}^{-3})\); however the SOC density of the SRC was significantly lower than that under grassland (Walter, Don and Flessa, 2015).

### 5.4.8 Afforested areas

Afforested areas maintain and increase carbon stocks in forests in above- and below-ground biomass, deadwood, litter, and soil. A major increase in the forest carbon reservoir is possible through afforestation of non-forest land.

The capacity of forests to store carbon varies strongly between regions in Europe. While young forests have initially high carbon sequestration rates, these decline in ageing forests. Mature forests may eventually reach an equilibrium at which relatively little further sequestration takes place (SFC, 2010).

However, if harvested then these areas become sources of emissions. In general, a forest stand acts as a carbon source for some years after final harvesting or thinning. Harvesting at small scales, retaining canopy cover and/or early reforestation can limit loss of carbon. Close-to-nature forestry with longer rotation periods maintains relatively higher soil-carbon stocks. Whole-tree harvesting can lead to losses of nutrients, and carbon losses in soil (particularly after stump extraction) and acidification unless appropriately compensated (e.g. through ash recycling). It is important to stabilize stands against biotic and abiotic disturbances to avoid large-scale loss of soil carbon (e.g. from drainage or wild fires) (SFC, 2010).

### 5.4.9 Catch crops / Green cover

Cover crops can mitigate GHG emissions in four main ways:

- Increase of soil organic carbon content.
- Decrease soil erosion during the fallow period.
- Reduction in N leaching (and subsequent N2O emissions)
- Reduction in the amount of N that needs to be applied to the following crop (reducing synthetic fertiliser use).

The repeated use of cover crops can help to maintain levels of soil organic carbon, though occasional use has little impact. See soil review for details of evidence.

The main on-farm mitigation effect of cover crops is via enhanced soil carbon storage. Soil organic carbon is increased by cover crops due to the increased duration of vegetation cover (Abdalla et al, 2014). Poeplau and Don (2015) used data from 139 plots at 37 different sites worldwide to determine that cover crops significantly increased SOC in comparison with reference croplands and that this increase in SOC persists beyond the length of the cover crop introduction – they found a mean annual SOC sequestration of 0.32 +/− 0.08 Mg ha\(^{-1}\) yr\(^{-1}\) to an average maximum increase of 16.7 Mg ha\(^{-1}\). Bulgarian studies carried out in test plots have reported that winter cover vastly reduces (by as much as 500 times) losses of SOC and nutrients in comparison with fallow periods (Flynn et al, 2007).

The effect on reducing N2O emissions through the uptake of nitrate and ammonium is less clear cut. Winter plant cover has the potential to reduce nitrous oxide emissions from soils by reducing the amount of mineral nitrogen stored in the soil. The impact of cover (or catch) crops on N loss by leaching, and the indirect N2O emissions that result from leachate is difficult to predict, and subject to significant uncertainty due to the number of site specific variables such as winter rainfall, existing soil N content and soil type. A review of the impact of cover crops on N2O emissions (Basche et al,
2014) reported that 60% of studies found that cover crops increased N2O emissions while 40% decreased them. Parkin et al. (2006; cited in Abdalla et al, 2014), state that ‘Rye cover crops accumulate significant proportions of applied N, greatly reducing the amount of NO3 lost in drainage water, reduce soil inorganic N levels, increase evapotranspiration, and reduce drainage losses and N2O emissions’. In contrast, however, other studies show that crop residues with low C:N ratios, e.g. legume crops, may increase N2O emissions (Toma and Hatano, 2007; cited in Abdalla et al, 2014). Gomes et al (2009) also stated that ‘It has been shown that cover crops can enhance soil nitrous oxide (N2O) emissions, but the magnitude of increase depends on the quantity and quality of the crop residues’. Li et al (2015) state that N2O emissions from legume based catch crops are similar to those from fallow land and non-legume based catch crops.

Sowing a catch/cover crop in the autumn before cultivation of a spring crop to reduce the fallow period reduces losses of N2O and soil nitrate (Sundermeier 2009, cited in Abdalla et al, 2014). Gomes et al (2009) state that an increase in N2O emissions is seen in the short term following cover crop management. This is supported by Brozyna et al (2013) who found that ‘periods of high N2O emissions coincided with cover crop and grass-clover residue turnover’. N2O emissions are generally greater following tillage in the spring (Brozyna et al, 2013; Li et al, 2015); however, peak N2O emissions occur at different times depending on the catch crop (Li et al, 2015). Higher emissions from cover cropped areas are also seen after freezing events (Petersen et al, 2011). However, the amount of N2O emissions stimulated following spring cultivation can be reduced through the use of reduced tillage (Petersen et al, 2011).

There can also be small reductions in CO2 if the cover crops lead to a reduction in the amount of synthetic fertiliser applied, but these are likely to be offset by the small increase in diesel used for cultivation of the cover crop (see Table 5) (MacLeod et al, 2015). The effects will depend on the type of crop used. Legume cover crops can suppress weeds (thereby avoiding emissions from additional herbicides that would have been required for their destruction), increase SOC, and reduce the amount of N fertilizer required for subsequent crops, as they are able to biologically fix N (Abdalla et al, 2014). Non-legume cover crops also bring benefits by taking up excess soil nutrients, and improving the structure of the soil (Abdalla et al, 2014).

The full potential for GHG emission reductions is largely dependent on how the crop is subsequently utilised – e.g. whether it is used in anaerobic digestion, used as a fertilizer or mulched.

The range of abatement rates for cover crops sown in the fallow period in arable cropping systems is shown in Table 5.

**Table 5: Abatement rates for cover crops sown during the fallow period for arable rotations**

<table>
<thead>
<tr>
<th>Mitigation effect</th>
<th>Abatement rate</th>
<th>Source</th>
<th>Location</th>
</tr>
</thead>
<tbody>
<tr>
<td>Increased soil C</td>
<td>0.874 +/- 0.393 tCO2e/h/yr</td>
<td>Pellerin et al. (2013) (based on Justes et al. 2012)</td>
<td>France</td>
</tr>
<tr>
<td></td>
<td>0.874 +/- 0.393 tCO2e/h/yr</td>
<td>Schulte et al. (2012)</td>
<td>Ireland</td>
</tr>
<tr>
<td></td>
<td>1.75 tCO2e/h/yr</td>
<td>Kirk et al. (2012)</td>
<td>United Kingdom</td>
</tr>
<tr>
<td></td>
<td>0.874 +/- 0.393 tCO2e/h/yr</td>
<td>Posthumus et al. (2013)</td>
<td>Ireland</td>
</tr>
<tr>
<td></td>
<td>0.874 +/- 0.393 tCO2e/h/yr</td>
<td>Pellerin et al. (2013)</td>
<td>United Kingdom</td>
</tr>
<tr>
<td>Reduce direct and indirect N2O Emission Factors</td>
<td>Highly variable</td>
<td>Pellerin et al. (2013)</td>
<td>France</td>
</tr>
<tr>
<td></td>
<td>0.46 tCO2e/h/yr</td>
<td>Schulte et al. (2012)</td>
<td>New Zealand</td>
</tr>
<tr>
<td></td>
<td>Leached N reduced by 30kgN/ha = 0.11tCO2e/ha</td>
<td>Cameron et al. (2002) (cited in O'Hara, 2003)</td>
<td>France</td>
</tr>
<tr>
<td>Reduce amount of applied N</td>
<td>0.08 tCO2e/h/yr</td>
<td>Pellerin et al. (2013)</td>
<td>France</td>
</tr>
<tr>
<td>Fieldwork CO2 – increased diesel use</td>
<td>-0.08 tCO2e/h/yr</td>
<td>Pellerin et al. (2013)</td>
<td>France</td>
</tr>
</tbody>
</table>

Source: (MacLeod et al, 2015)
5.4.10 **NITROGEN-FIXING CROPS**

The inclusion of legumes in arable rotations and grass swards can mitigate GHG emissions in three main ways:

- Reducing or eliminating the need for mineral N fertilizers on the fields where the legumes are grown reduces direct emissions from N fertilizers.
- The breakdown of legume residues releases N over the following growing season and reduces the amount of fertilizer-N that needs to be applied to the following crop.
- By reducing the need for mineral N fertilizers, GHG emissions from fertilizer manufacture are also reduced.

Including clover in grassland was considered to potentially reduce GHG emissions by 15 to 32% in Scotland (Feliciano et al, 2013).

MacLeod et al (2015) provide an overview of the abatement rates identified for legumes in arable systems (as well as on grassland) in various studies (see Table 6).

**Table 6: Abatement rates for legumes in arable systems and on grassland**

<table>
<thead>
<tr>
<th>Mitigation effect</th>
<th>Abatement rate (AR)</th>
<th>Source</th>
<th>Location</th>
</tr>
</thead>
<tbody>
<tr>
<td>Reduction in direct and indirect N₂O</td>
<td>Grain legumes 1.02 tCO₂e/ha/yr</td>
<td>Pellerin et al. (2013)</td>
<td>France</td>
</tr>
<tr>
<td></td>
<td>Forage legumes 0.17 tCO₂e/ha/yr</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Reduced fertiliser production</td>
<td>Grain legumes 0.55 tCO₂e/ha/yr</td>
<td>Pellerin et al. (2013)</td>
<td>France</td>
</tr>
<tr>
<td></td>
<td>Forage legumes 0.16 tCO₂e/ha/yr</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Total reduction in GHG*</td>
<td>Grain legumes 1.99 tCO₂e/ha/yr</td>
<td>Pellerin et al. (2013)</td>
<td>France</td>
</tr>
<tr>
<td></td>
<td>Forage legumes 0.33 tCO₂e/ha/yr</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Reduction in direct and indirect N₂O</td>
<td>Clover 0.5 tCO₂e/ha/yr</td>
<td>Moran et al. (2008, p43)</td>
<td>United Kingdom</td>
</tr>
<tr>
<td>Reduction in on-farm GHG</td>
<td>Substituting legumes for cereals: 1.19 tCO₂e/ha/yr (barley) to 2.20 tCO₂e/ha/yr (maize)</td>
<td>Decuquet and Moran (2015)</td>
<td>France</td>
</tr>
</tbody>
</table>

*Including minor changes not specified in table.

Source: (MacLeod et al, 2015)

5.4.11 **CREATION OF "BUFFER ZONES" FOR HIGH NATURE VALUE AREAS, NATURA 2000 OR OTHER BIODIVERSITY PROTECTION SITES, INCLUDING ALONG HEDGEROWS AND WATER COURSES**

No particular benefit for GHG emission reductions found in the literature.

5.4.12 **BORDERS, IN-FIELD STRIPS AND PATCHES MANAGED FOR WILDLIFE OR SPECIFIC FAUNA (HERBACEOUS BORDER, PROTECTION OF NESTS, WILDFLOWER STRIPS, LOCAL SEED MIX, UNHARVESTED CROPS)**

No particular benefit for GHG emission reductions found in the literature.
5.4.13 **Production on arable land with no use of fertiliser (mineral fertiliser and manure) and/or plant protection products, and not irrigated, not sown with the same crop two years in a row and on a fixed place;**

No specific evidence was found for this particular practice, although the effect of not using fertilisers should lead to reduced levels of the N$_2$O emissions that would have been associated with their use.

5.4.14 **Conversion of arable land into permanent grassland extensively used.**

See the section on permanent grassland measure above.
6 LITERATURE REVIEW ON THE EFFECTS ON CLIMATE ADAPTATION OF FARMING PRACTICES RELEVANT TO THE PILLAR 1 GREENING MEASURES

6.1 INTRODUCTION

In relation to climate adaptation, there are three main types of adaptation measures that are relevant at the farm level (OECD, 2010):

- Those that reduce the vulnerability of affected agro-ecosystems and agricultural soils;
- Those that reduce the exposure of a farming system to the effects of climate change such as drought, heavy rainfall, and storms by hazard management; and
- Those that increase resilience, both in ecosystems by conserving resources and the resilience of the farming population to enable them to overcome the losses that do occur.

The main pressures on agriculture, as a direct result of climate change, are water availability due to reduced rainfall and droughts in some areas as well as increased rainfall, storms and flood events in others, overall temperature variations, increased wind damage to crops, presence and persistence of pests and diseases, as well as fire risks. Soil erosion is a key climate change impact in the agriculture sector. Different studies show that higher precipitation will force a higher run-off and soil loss. Lee et al. (1996) argue that 20% more precipitation would lead to 37 to 40% higher run-off and soil loss. (Moret et al, 2006) calculate that 10-20 % higher precipitation would lead to a 300 percent increase of soil loss and run-off. Droughts can also increase soil erosion – wind and water erosion. Both effects are thought likely to increase in frequency and intensity in a number of EU regions.

Many adaptive actions have been identified as having the potential to strengthen the resilience of farms and agroecosystems as well as reducing vulnerability (Bindi and Olesen, 2011; Hjerp et al, 2012; Iglesias et al, 2007; Smith and Olesen, 2010). These sources have been the primary ones used for this literature review.

Although a number of studies highlight a range of farming practices that have this potential, details on factors influencing their effectiveness or the scale of their impact are less common. In addition, many of the adaptation measures identified are also identified as beneficial for soils (e.g. by reducing erosion, increasing the infiltration capacity of water into the soils and reducing flooding) and in these cases the literature review cross refers to the soils literature review.

6.2 CROP DIVERSIFICATION GREENING MEASURE

6.2.1 DIVERSIFICATION OF CROPS

Growing two or more crops in proximity in the same field during a growing season in order to promote interaction between them can increase the resilience of crops against pests and diseases (Hjerp et al, 2012). Kremen and Miles (2012) found that compared with conventional farming systems, diversified farming systems support substantially greater biodiversity, soil quality, carbon sequestration, and water-holding capacity in surface soils, energy-use efficiency, and resistance and resilience to climate change. Relative to conventional monocultures, diversified farming systems also enhance control of weeds, diseases, and arthropod pests and they increase pollination services;

6.2.2 CROP ROTATIONS

Crop rotations can help increase the resilience of cropping systems to climate change by reducing the amount of water required to irrigate crops as well as increasing the resilience of crops to pests and disease. However, the nature of the crop rotation will determine its effect, for example the choice of crops (e.g. in relation to its water needs) and whether the soil is under cover to avoid evapotranspiration (BIO Intelligence Service, 2010). The BIO IS study suggests that rotations performing well without irrigation, such as oilseed rape/wheat or legume/wheat/cereal will contribute to water savings.
6.2.3 **WINTER SOIL COVER**

*See EFA section*

6.2.4 **CATCH CROPS**

*See EFA section*

6.3 **MAINTENANCE OF PERMANENT GRASSLAND GREENING MEASURE**

Permanent grasslands have proven effective in mitigating threats from floods through their high capacity in subsurface storage. Wet grasslands in particular can serve as a buffer zone for agricultural runoff and contribute to reducing erosion. The relevance of this measure increases with the severity of climate change impacts (Hjerp et al, 2012).

6.3.1 **PLOUGHING IN ONE AREA AND CONVERTING LAND TO PG IN ANOTHER**

Ajayi and Horn (2016) found that permanent grassland, established from degraded arable land aged 1, 8 and 19 years, caused a significant improvement in pore structure, results increasing with the age of the grassland. This resulted in improved water retention, air and water fluxes. Thus suggesting that the conversion of land to grassland could significantly reduce the vulnerability soil as the quality is increased.

6.4 **ECOLOGICAL FOCUS AREAS**

6.4.1 **LAND LYING FALLOW**

No specific literature on the benefits of fallow land for climate adaptation was found.

6.4.2 **TERRACES**

Terracing on sloping land is important for reducing soil erosion and surface run-off. As such they can reduce the risk of flooding. For further information see the soils literature review.

6.4.3 **LANDSCAPE FEATURES**

6.4.3.1 **Hedgerows and wooded strips**

Hedgerows can contribute to alleviating many of the risks associated with climate change, for example reducing wind erosion, protecting crop areas, and providing shelter for livestock. Hedgerows located across the slope can substantially reduce the run off of water, hence helping reduce the effects of high rainfall and flood events.

They can also help species adapt to climate change by providing connectivity within fragmented landscapes thereby facilitating the movement of species between isolated remnants of habitat which may be necessary as species adapt to climate change (Cumulus Consultants, 2007).

6.4.3.2 **Isolated trees**

In isolation there is no particular evidence of the benefits of trees for climate adaptation, although they may provide shelter for livestock if the canopy is sufficiently broad.
6.4.3.3 Trees in a line

Shelterbelts can contribute to alleviating many of the risks associated with climate change, for example reducing wind erosion, the protection of crops and they can also provide shelter for livestock.

Shelterbelts, positioned across the slope can substantially reduce the run off of water, through enhancing water absorption hence helping reduce the effects of high rainfall and flood events. The soil literature review cites the Pontbren study in Wales (UK). Although the study was carried out on intensively management permanent grassland and therefore not directly relevant to the EFA measure, this showed that the soils under these shelterbelts had infiltration rates as much as 67 times greater than those in the adjacent intensively managed pasture. Tree roots were found to be directly creating channels for infiltration, with deep rooted species such as ash (*Fraxinus excelsior*) particularly effective in this respect (Keenleyside, 2012).

6.4.3.4 Trees in groups and field copses

No specific literature was found on the climate adaptation benefits of this landscape feature. However, the main benefit is likely to be in providing shelter for livestock.

6.4.3.5 Field margins

See buffer strips

6.4.3.6 Ponds

No specific literature was found on the climate adaptation benefits of this landscape feature.

6.4.3.7 Ditches

No specific literature was found on the climate adaptation benefits of this landscape feature.

6.4.3.8 Traditional stone walls

Traditional stone walls on sloping land can be important for reducing soil erosion and surface run-off. Where situated on sloping land can help to preserve soil productive capacity, maintain landscape values, reduce risk of flooding and reduce pressures on water quality.

6.4.4 Buffer strips

Strips of vegetation, including strips of grassland along water bodies and grass margins (as well as hedgerows, field corners and trees as mentioned above) act as buffer zones or a shield against overland flow from agricultural fields and reduce run-off from reaching the watercourse, thus decreasing erosion and the movement of pollutants into watercourses. Vegetation strips have been shown to be a highly effective measure that can almost completely offset run-off and erosion in permanent crops, e.g. as almonds and olives (Gómez et al, 2009; Martínez Raya, Durán Zuazo and Francia Martínez, 2006). They can also be combined with other adaptation options such as winter cover to enhance the effect. However, other research has shown that the design of the buffer strip (width, location, vegetative cover and ensuring these are matched to soil type) are important factors that determine the effectiveness of buffer strips and field margins in this regard (see soil literature review).

Buffer strips and field margins, depending on how they are managed may also provide wildlife corridors which can help improve the connectivity of habitats, thereby facilitating the movement of species which may be necessary as species adapt to climate change (Cumulus Consultants, 2007). See also the biodiversity literature review.
6.4.5 **Agroforestry**

Agro-forestry can contribute to alleviating many of the risks associated with climate change, for example reducing wind erosion, protection of crops, and providing shelter for livestock. Agro-forestry systems are considered to be more resilient to the effects of climate change (e.g. increasing temperatures or droughts, water and biotic stresses and more extreme events) than monocultures of crops (see for example the SAFE: Silvoarable Agroforestry for Europe project in Montpellier, France\(^{26}\)). Effects of agroforestry in the SAFE project related to climate adaptation included: trees providing shelter to crops and reducing damage resulting from high spring temperature; more diverse crops helped combat pest and diseases and has the potential to improve pollinators.

Louwagie, Gay and Burrell (2009c) also found that agro-forestry systems are beneficial in increasing the water holding capacity of the soil and can therefore help protect against climate variability (see also (Béduneau and Gabory, 2012a). This quality means that these systems may have a function in helping reduce flood risk. Evidence also shows the benefits of these systems in reducing soil erosion (see soil literature review).

6.4.6 **Strips of Eligible Hectares Along Forest Edges**

No specific literature was found on the climate adaptation benefits of this practice, but the benefits are likely to be similar to those identified for buffer strips.

6.4.7 **Short Rotation Coppice**

No specific literature was found on the climate adaptation benefits of this practice.

6.4.8 **Afforested Areas**

Targeted planting can interrupt pollutant pathways as well as assist in sediment control, and flood alleviation, hence helping address issues such as flooding, flash floods, storms and resulting soil erosion. For the measure to be beneficial and not have negative effects on other environmental objectives (especially biodiversity), species planted must be native and planting has to be situated in appropriate locations.

6.4.9 **Catch Crops / Green Cover**

Winter plant cover helps prevent nutrient run off and leaching of N and P and pesticides into water sources. It also initiates processes for soil quality improvement and erosion control. It is very relevant to climate adaptation given the expected increase in extreme precipitation events, and changing precipitation patterns (milder and wetter winters). The measure promotes soil fertility and enhances infiltration, which reduce losses of nutrients and the possibility of groundwater contamination as well as protecting against the future increased risk of flooding (Frelih-Larsen et al, 2014). Plant cover in winter can improve soil structure by increasing the amount of organic matter in the topsoil, which decreases silting. By slowing down wind speeds at ground level plant cover thus also reduces wind erosion.

For climate adaptation purposes this measure is of most value in countries where rainfall is expected to increase and/or storm events are expected to increase in frequency. This tends to be northern, central and eastern Member States. However, it could exacerbate problems in countries with water scarcity issues as further cropping would increase these problems (e.g. countries like Spain, Italy, Greece, Bulgaria, Cyprus, Malta, Portugal) (Hjerp et al, 2012).

Cover crop mixes could provide multifunctional adaptation to both extreme rain events and drought as grasses are best for decreasing erosion, legumes reduce the risk of yield reductions in subsequent crops, and brassicas have deep taproots that break through compacted soil and decrease soil compaction (Kaye and Quemada, 2017).

See the soil literature review for further information.

6.4.10 Nitrogen-fixing crops.
No specific literature on the climate adaptation benefits of N-fixing crops has been found, except through their use in a crop rotation (see above).

6.4.11 Creation of "buffer zones" for high nature value areas, Natura 2000 or other biodiversity protection sites, including along hedgerows and water courses.
No specific literature on the climate adaptation benefits of such zones was found, but they may help improve the connectivity of habitats, thereby facilitating the movement of species which may be necessary as species adapt to climate change.

6.4.12 Borders, in-field strips and patches managed for wildlife or specific fauna (herbaceous border, protection of nests, wildflower strips, local seed mix, unharvested crops)
See section on buffer strips and field margins.

6.4.13 Production on arable land with no use of fertiliser (mineral fertiliser and manure) and/or plant protection products, and not irrigated, not sown with the same crop two years in a row and on a fixed place;
No specific literature was found on the climate adaptation benefits of this practice, although the rotation of crops will help increase the resilience of the system against pest and disease (see crop diversity and rotation sections above).

6.4.14 Conversion of arable land into permanent grassland extensively used.
Covered under the permanent grassland measure.
7 LITERATURE REVIEW ON THE EFFECTS ON AMMONIA OF FARMING PRACTICES RELEVANT TO THE PILLAR 1 GREENING MEASURES

7.1 INTRODUCTION

Ammonia (NH₃) emissions contribute to the formation of secondary particulate matter (PM) and tropospheric ozone, both with serious impacts on air quality. Ammonia also contributes to nitrogen deposition and acidification of ecosystems across Europe (EEA, 2010b). Ammonia emissions from agriculture in Europe are increasing (see CAP CMEF context indicators).

Ammonia may be produced from manure or fertiliser applied to farmland, or from urine and dung. Most of the ammonia from farmland originates from animal manure or slurry in animal houses, manure or slurry storage systems and following the application of animal manure or slurry to agricultural land (Oenema et al, 2007). Urea fertilizers and fertilised crops also contribute to ammonia emissions (Oenema et al, 2007). Other biomass such as crop residues added to farmland produces very little ammonia so is not considered in this review.

This literature review focuses on ammonia production from the following agricultural practices on land associated with greening:

- manure or slurry application to permanent grassland; and manure or slurry application to cropland subject to crop diversification and/or EFA and to any other EFA elements;
- urea fertiliser application to grassland and cropland subject to crop diversification and/or EFA, and to any other EFA elements;
- grazing practices that might influence urine and dung production on grassland or EFAs.

Ammonia is also produced by volatisation from stored manure and slurry and animal housing, but this pathway is not considered in this review, as these practices are very unlikely to be affected by greening.

Due to the fact that ammonia emissions are primarily defined by agricultural practices rather than cropping systems or landscape elements, the impact of greening on ammonia emissions is through the indirect effects of changes in cropping systems or EFA landscape elements on these agricultural practices. Fertilisers are most likely to be applied to arable land, as fertiliser use on permanent grassland is a less frequent practice in the EU. Conversely, manure and slurry are most likely to be applied to permanent grassland or temporary grassland, as most farms with livestock housing are dairy farms with most land under grassland rather than arable crops. However, there are plenty of exceptions to these generalisations, and therefore this literature review focuses on the evidence for agricultural practices and ammonia rather than the greening measures directly.

The degree to which a farmer follows good practice in application of manure or slurry to agricultural land is not expected to be greatly influenced by greening (other than particular rules, for example the Scottish nutrient balance requirement). Manure is not always spread on the producer’s own land, therefore any impacts of the greening measures on the density of grazing animals, and thus on the amount of manure or slurry they produce whilst in housing, is indirect and was not considered in this review. As a result of the lack of literature found regarding greening specific measures some subheadings have not been included in this section. There is an additional section of ammonia mitigation information.

27 manure type can be categorized as either slurry, consisting of mixed urine, faeces and water with very little bedding material and with a dry matter content typically in the range 1-10%, or as farm yard manure (FYM) consisting of urine and faeces mixed with large amounts of bedding material (typically straw) having higher dry matter content.
7.2 General Information

7.2.1 Ammonia Emissions from Manure or Slurry Application to Farmland

Most manure or slurry produced by livestock in housing is spread on agricultural land. A wide range of equipment and techniques are used to spread slurry and solid manure to land, and emissions of ammonia vary greatly with different techniques and machinery. Ideally, through the application of manure or slurry at the moment when the crops require the nutrient and adjusting the rates to the crop needs, it is possible to improve the crop yields, reduce fertilizing costs, and enhance the levels of soil organic matter (Bernal et al., 2015). Conversely, it is possible to lose most of the nitrogen through volatisation of ammonia. If the active nitrogen is converted to NH4+, it will adsorb to clay in the soil and be available for plant root uptake or converted by nitrifying bacteria into nitrate or nitrous oxide, instead of being volatised as ammonia. The techniques that reduce ammonia emissions may increase the likelihood of nitrate leaching and/or nitrous oxide emissions from nitrification of ammonium in the soil, but this is not always the case (Webb et al., 2010).

Factors that influence ammonia emissions from manure or slurry application are:

Livestock type and form of waste: Slurries typically contain 40-80% of the N in the ammonium form with the remainder as organic N and none as nitrate, whilst farm yard manure typically contains a much lower proportion of the N in the ammonium form and may contain a small fraction in the nitrate form28. Pig manure typically has a higher total N and available (mineral) N content than cattle manure but this depends on water content. Both poultry litter and dry poultry manure contain higher total N contents than cattle or pig manures and between 30-50% of the total N may be in an inorganic form as uric acid or ammonium.

Acidification: The acidification of slurry by mixing in acids29 before field application converts more of the nitrogen to NH4+, thereby reducing the potential for ammonia emissions30. However, acidified slurry may produce more nitrous oxide emissions, as shown in a laboratory study using pig slurry (Fangueiro et al., 2010).

Machinery: High levels of ammonia emissions result from machinery which spreads slurry or solid manure over the whole soil surface (‘broadcast’) by throwing it into the air (Bernal et al., 2015). Low efficiency methods of slurry and manure application with medium levels of ammonia emissions include slit injection, trailing shoe, slurry dilution, and band spreading for liquid slurry (Klimont and Brink, 2004). High efficiency methods of slurry and manure application with low levels of ammonia emissions involve the deep and shallow injection of liquid manure (Klimont and Brink, 2004).

Soil incorporation: Most ammonia volatilises within the first 12 hours, so speed of incorporation into the soil reduces ammonia emissions considerably. Ammonia emissions are much lower if slurry is incorporated into soil within four hours after application, or solid manure is incorporated within 12 hours after application, than by ploughing into the soil the day after application (Klimont and Brink, 2004). Even short (4–6 h) delays in incorporating manures after application will greatly reduce the efficacy of rapid incorporation as a means of NH3 abatement (Webb et al., 2010). On grassland, soils saturated by winter or spring rainfall or high water tables incorporate slurry more slowly than dry soils, resulting in higher ammonia emissions, although lower soil temperatures reduce volatisation (Bourdin et al., 2014).

Season and weather:

Climatic and soil conditions that enhance grass growth reduce NH3 and GHG emissions due to the more rapid uptake of nitrates and ammonium from the soil, as shown by significantly lower NH3 and GHG emissions from spring application of cattle slurry to grassland in Ireland compared to summer

29 nitric and sulfuric acid with liquid manure and aluminium sulfate with solid manure
Ammonia emissions from grassland are increased by higher temperatures and higher wind speeds (Sommer et al., 2003), which correspond to summer climate conditions.

**Soil type and soil conditions:**
Grassland soils with high cation exchange capacity, because of high clay content and/or because of high soil organic matter content, are less prone to ammonia volatilization from manure or slurry than soils with low cation exchange capacity (Saggar et al., 2004). Soil temperature, pH, moisture content, organic matter content, nitrogen content, and clay content all affect the presence of ammonia-oxidising bacteria (and archaea) in soils (Stempfhuber et al., 2014). High presence and activity of the bacteria in soil decreases the likelihood that manure and slurry applications will produce ammonia emissions rather than nitrates in the soil water and/or nitrous oxide emissions. However, the relationship between soil conditions, activity of ammonia-oxidising bacteria, and ammonia emissions is influenced by many factors and it is not possible to make general predictions (Stempfhuber et al., 2014).

### 7.2.2 Ammonia emissions from fertiliser use

Factors that influence ammonia emissions from fertiliser application to farmland are:

**Substitution of urea fertilizer and ammonium carbonate for other mineral N fertilisers:** The proportion of N lost as NH₃ is generally higher for urea than for other mineral N fertilisers (ammonium nitrate, calcium ammonium nitrate, calcium nitrate and urea ammonium nitrate). However, these fertilisers are almost always more expensive than urea.

**Efficient application of fertiliser (machinery, tillage and additives):**
The risk for high NH₃ losses from urea fertiliser may be reduced by incorporation of the fertilizer into the soil and the use of urease inhibitors together with fertiliser. The risk for high ammonia losses from urea fertiliser may be reduced by applying the urea to the soil surface beneath the crop canopy, taking advantage of the relatively low wind speed within well-developed crop canopies, reducing the rate of vertical NH₃ transport and increasing foliar NH₃ absorption (Sommer, Schjoerring and Denmead, 2004). However, soil incorporation in growing crops and fertiliser application to the soil surface beneath the crop canopy require suitable machinery.

**Soil and weather conditions:**
The risk for high NH₃ losses from urea fertiliser may be reduced by the selection of acidic fertilizers on calcareous soils, and use of fertilizers with a high content of carbonate-precipitating cations. Volatilization of surface-applied urea increases linearly as soil water content increases, until the soil reaches saturation, particularly if application is followed by a drying period (Jones et al., 2013). The timing of application in relation to weather conditions is a critical factor. The risk of ammonia volatilisation occurring is increased by light rain (or heavy dew) followed by a dry period, warm temperatures and windy conditions (Jones et al., 2013). Heavy rainfall after application to dry soil will result in penetration into deeper soil where ammonia volatilisation does not occur.

**Vegetation cover or crop residues:**
Vegetation cover or crop residues generally increase ammonia volatilization from urea fertiliser (San Francisco et al., 2011), because the urease enzyme necessary for hydrolysis is produced by microorganisms that are much more active in surface residue than in mineral soil; crop residues often have higher pH than soil, which increases ammonia in solution; residue may locally increase moisture, which also increases ammonia in solution and available for volatilization; and residue can prevent N from moving into soil (Soane et al., 2012).

### 7.2.3 Ammonia emissions from livestock urine and dung on grassland

Ammonia emissions arise principally from urine, as dung contains mostly organic N forms, which will subsequently mineralize at a rate dependant on soil and environmental factors, whereas N in urine is predominantly in an inorganic form and immediately susceptible to losses via ammonia volatilization, leaching and denitrification (Selbie, Buckthought and Shepherd, 2015).
Factors that influence ammonia emissions from urine and dung are:

- **Type of grazing livestock, length of grazing period and stocking density**: Cattle dung contains much less nitrogen content than sheep or goat dung but there is more of it, whilst there is also large variation between dairy and beef cattle. On the other hand, cattle are less likely to be grazing all year round and so annual emissions from grazing livestock (excluding emissions from manure or slurry in livestock housing) may not differ greatly (Newell Price et al, 2011).

- **Grassland fertilisation / supplementary animal feeding**: For livestock which receive supplementary feeding in the field, the N content of the animal feed has an influence on the ammonia emissions from their excrement in the field. Cattle fed with excess protein have much higher N content in their urine and therefore produce higher ammonia emissions (Cole et al, 2005). Conversely, cattle with a high roughage diet, for example through grazing on unfertilised pasture, have much lower ammonia emissions from urine than cattle fed on intensively fertilised high protein grass (Külling et al, 2003). Sheep feeding on clover rich pasture are likely to produce higher ammonia emissions than on grassland with no legumes.

- **Grassland soil compaction or waterlogging**: Grassland soil compaction reduces the rate of incorporation of urine into the soil, which increases the likelihood that active nitrogen is volatised as ammonia or nitrous oxide rather than being incorporated into the soil as ammonium or nitrate. Waterlogged soil also reduces the rate of incorporation of urine, but also increases the likelihood that active nitrogen is dissolved or washed away directly.

### 7.3 Crop Diversification Greening Measure

This section considers only the way in which crop diversification might change factors that influence ammonia emissions, but it should be noted that a number of the key factors listed above are not addressed because they are not influenced by greening (e.g. the type of fertiliser applied, whether manure or slurry are applied to arable land or not, machinery used).

#### 7.3.1 Diversification of Crops

The increased use of legume crops can be expected to reduce ammonia emissions if farmers do not apply fertilisers or manure or slurry to the legume crop and additionally if they reduce fertiliser use on subsequent crops. If crop diversification into legume crops increases soil organic matter, this could also indirectly decrease ammonia emissions (see above).

As most fertilisers are applied before crops are planted or very soon after planting (basal application), the influence of crop diversification on the likelihood of ammonia emissions from fertiliser application is most likely to be through the change in the season and timing of crop sowing and associated changes in tillage practices. It is not possible to make any generalisations about how this might affect ammonia emissions.

Ammonia emissions from fertiliser applications applied to growing crops are influenced by the crop structure (see above), so ammonia emissions from top up fertiliser applications to different crops will differ. No current literature was found to document this.

#### 7.3.2 Catch Crops

Cover crops generally increase ammonia volatilization if urea fertiliser is applied directly on the crop (see above); however, this is rarely done. If the use of cover crops is combined with a change to minimum tillage and direct drilling, the presence of crop residues on the soil surface can increase ammonia emissions from fertiliser applications, but also later in the season the increased soil organic matter content can reduce ammonia emissions (see above). However, nitrogen-fixing crop residues left on the soil surface as green manure can produce ammonia emissions (Janzen and McGinn, 1991).

### 7.4 Maintenance of Permanent Grassland Greening Measure

Ammonia emissions from permanent grassland may be lower or higher compared to ammonia emissions from arable land, depending on the intensity of manure or slurry application and associated ammonia emissions on the former compared to the intensity of ammonia-producing fertiliser
applications on the latter. It is therefore not possible to state that the prevention of grassland conversion or vice versa will increase or reduce emissions.

Ammonia emissions from fertiliser or manure or slurry are affected by grassland soil structure, particularly by compaction, soil organic matter content, pH and temperature, weather conditions, timing and machinery used for application, as well as the type of fertiliser or livestock (see above).

Ammonia emissions on grassland from livestock urine and dung may be affected by the type of grazing livestock, length of grazing period, stocking density, supplementary feeding of livestock, and grassland soil structure (see above), and indirectly by the influence of fertiliser use on grass protein content.

However, the maintenance of permanent grassland measure does not directly influence any of these farming practices unless extra conditions have been imposed which might change management practices on grassland (e.g. the requirement for a nutrient management plan in Scotland).

7.5 ECOLOGICAL FOCUS AREAS

7.5.1 LANDSCAPE FEATURES

7.5.1.1 Hedgerows and wooded strips

See trees section below.

7.5.1.2 Isolated trees/Trees in a line/Trees in groups and field copses

The introduction of trees around or within fields can increase soil infiltration capacity, which can reduce ammonia emissions if fertiliser is applied to dry soil followed by rain. It may also reduce the likelihood that soil becomes waterlogged, and this could slightly decrease the possibility of ammonia emissions from fertiliser applied directly to waterlogged soils. However, it is not likely that fertiliser will be applied in these weather conditions due to the risk of damage to the soil.

Lines, groups and even single trees provide a slight canopy over crops which the potential to catch some of the vertical ammonia emission from fertiliser applications, allowing it to be converted to nitrate, with a small amount absorbed by the leaves. A modelling study (Dragosits et al, 2006) indicated that tree belts were able to reduce levels of ammonia deposition from manure applied to nearby areas, with higher effectiveness observed if the trees surrounded the sensitive crop areas as opposed to the sites of application. Another modelling study (Asman, 2008) found that shelterbelts were effective in capturing amounts of dry deposited gaseous ammonia, the closer to the source and taller the trees the higher the effectiveness. Ammonia concentrations in the air were lower following the introduction of potted trees downwind from poultry house fans in the US (Patterson et al, 2008).

7.5.1.3 Field margins

See buffer strips.

7.5.2 BUFFER STRIPS

Buffer strips with tall vegetation may capture ammonia emissions by acting as wind buffers (see trees above).

7.5.3 AGROFORESTRY

Trees on grassland grazed by livestock can capture a significant proportion of the ammonia emissions from dung and urine, and return it to the soil as nitrate (Bealey et al, 2016).

7.5.4 AFFORESTED AREAS

A modelling study (coupled disposition and turbulence model) predicted that afforestation surrounding farm slurry lagoons can recapture up to 19% of ammonia emissions, and afforestation around under-storey systems (free-range poultry areas) could reduce emissions by up to 40% (Bealey, 2016). Another study (Bealey et al, 2016) concluded that afforesting areas downwind of livestock housing produced significant reductions in ammonia emissions on semi-natural habitats.
7.5.5 CATCH CROPS / GREEN COVER

If catch crops are incorporated into the soil as green manure to increase soil organic matter and soil fertility, and if this results in a reduction in fertiliser use on subsequent crops, this will indirectly reduce the likelihood of ammonia emissions.

7.5.6 NITROGEN-FIXING CROPS

The use of nitrogen fixing crops will have a significant indirect reduction effect on ammonia emissions if they result in a decrease in fertiliser or manure applications on the cropped area. However, nitrogen-fixing crop residues left on the soil surface as green manure can produce ammonia emissions (Janzen and McGinn, 1991).

7.5.7 PRODUCTION ON ARABLE LAND WITH NO USE OF FERTILISER (MINERAL FERTILISER AND MANURE) AND/OR PLANT PROTECTION PRODUCTS, AND NOT IRRIGATED, NOT SOWN WITH THE SAME CROP TWO YEARS IN A ROW AND ON A FIXED PLACE;

Production without using fertiliser of any sort will cut out the majority of agricultural ammonia emissions from arable land.
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Grades Doktor der Agrarwissenschaften, Agrar- und Umweltwissenschaftliches Fakultät, Universität Rostock.


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