Potential impacts of bioenergy developments on habitats and species protected under the Birds and Habitats Directives

ANNEX REPORT – EXTENDED ANALYSIS –
REVIEW AND ANALYSIS OF THE POTENTIAL DIRECT AND INDIRECT IMPACTS OF BIOENERGY DEVELOPMENTS ON HABITATS AND SPECIES PROTECTED UNDER THE BIRDS AND HABITATS DIRECTIVES

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1 Introduction

1.1 Scope and aims of the analysis

The aim of this assessment is to review and analyse the potential impacts of bioenergy developments on habitats and species protected under the Birds and Habitats Directives, by scoping the key risk factors associated with bioenergy and semi-quantifying the main potential impacts on EU protected habitats and species[1].

In particular, the **scope of this assessment** covers habitats and species of EU interest[2], which for this study comprise:

- 'natural habitats of Community interest' listed in Annex I of the Habitats Directive – hereafter referred to as HD habitats;
- 'species of Community interest' as listed in Annexes II and/or IV or V of the Habitats Directive – hereafter referred to as HD species; and
- bird species listed in Annex I of the Birds Directive that require special conservation measures – hereafter referred to as BD birds.

Strictly speaking, all bird species are protected under the Birds Directive, but to focus on those that are of high conservation importance this study is limited to those on Annex I. Migratory bird species which are not listed as Annex I are also not covered, as whilst they are also subject to similar special measures, they are not particularly susceptible to the impacts of bioenergy, and including them would give disproportionate emphasis to impacts on birds when other taxa are equally affected by bioenergy production. The review covers potential impacts on EU protected species and habitats within Europe, whilst taking into consideration that migratory species are subject to cumulative impacts along their whole migration routes and at wintering sites.

The review covers the main types of biomass used i.e. wood, biogas feedstocks and biofuel feedstocks and foreseen bioenergy development in line with the EU climate and energy targets. It considers the question of scale of use and the consequences for habitats in question. The analysis focuses on:

- the impacts associated with land-use change driven by crops used for energy, such as establishment of dedicated energy plantations or cultivation of dedicated energy crops and links to the conversion of permanent grassland and other semi-natural habitats to cropland (e.g. to maize for biogas);
- land management impacts (not including land-use change) of increased biomass demand (e.g. increased forest harvest), including:
    - i) more intensive removal of biomass from forest that would anyway be harvested, by taking leaves, branches, dead wood, extracting stumps, etc.;
    - ii) harvesting forests more frequently, thus reducing rotation age and the share of old forests and trees, increasing disturbance in the landscape;

[1] This report was written between March 2018 and February 2019, therefore newer data sources published since then have not been taken into account.
[2] The word species is used to refer to the species or subspecies taxa as listed in the directives.

- iii) harvesting forest that would otherwise not be harvested (as it would be uneconomical without energy subsidies); and
- iv) intensification of other land uses including arable and grasslands (e.g., taking more residues, more fertilisation, higher share of monoculture) in line with land use change predicted in key studies such as Recebio (Forsell et al, 2016a).

The assessment also identifies, but does not quantify, potential impacts associated with the physical construction and infrastructure needs of bioenergy plants and the processing biomass for energy (e.g. pellet mills, sawmills, storage facilities for biomass for energy). However, as such developments will be subject to appropriate assessments and mitigation measures that address other energy related built developments etc, these aspects are dealt with under another report produced in the project[3]. The scope of this study excludes impacts of transportation and end use of bioenergy (e.g. wood pellet heating, use of transport fuels), as well as aquatic vegetation-based biomass such as algae and bacteria. However, the scoping will examine whether it would be important to take account of within-habitat infrastructure such as forest roads (this differs from other renewable energies).

The following types of bioenergy feedstock are assessed:

1. Conventional crops (i.e. food and feed or forage crops)
    a. Crops for biofuel or biogas:
        i. Oilseed rape
        ii. Maize
        iii. Wheat or other cereals
        iv. Sugar-beet
        v. Grass
        vi. Others
    b. Residues from agriculture;
        i. Residues from crops (straw & stubble etc)
        ii. Residues from semi-natural habitat management

2. Energy crops on agricultural land
    a. Miscanthus and Reed Canary Grass etc
    b. Short-rotation coppice (SRC)
    c. Afforestation with fast growing trees
    d. Small-scale tree planting

3. Forest biomass from existing forest land
    a. Roundwood
    b. Residues (stumps, thinning, tops and branches, deadwood, rejected saw logs and other low value trees including deadwood).

[3] Project: "Reviewing and mitigating the impacts of renewable energy developments on habitats and species protected under the Birds and Habitats Directives" for European Commission under EC Contract ENV.D.3/SER/2017/0002

The review considers and attempts to quantify the following types of potential impacts from the above feedstocks on biodiversity components:

- Habitat loss from the physical land footprint of the energy technology (e.g. bioenergy plants and associated infrastructure, e.g. increased access roads for forestry) - briefly.
- Impacts on species from habitat loss and changes in land use (e.g. conversion of grassland to bioenergy crop).
- Direct mortality of species (e.g. losses of ground nesting birds during the harvest of bioenergy crops or through changes in intensity of harvest). Disturbance of species (e.g. from noise or visible movements of machinery or people such as during forestry operations).
- Habitat condition changes due to change in land management associated with biomass feedstock demand (e.g. increased fertiliser use, cultivations, more intensive forest management) or as a result of local pollution (e.g. nutrient flows in to rivers, dust or sediment deposition or surrounding vegetation).

It is important to note that there are impacts of bioenergy feedstock production on biodiversity beyond the scope of EU protected habitats and species. Furthermore, as a result of the nature of bioenergy production and reliance on feedstock use and global trade in forest and agricultural commodities a large proportion of the feedstock used within the EU come from beyond its borders (Forsell et al, 2016a), and there is strong evidence that this is having substantial impacts on biodiversity (Bowyer, 2010a; Meletiou et al, 2019). However, these issues are beyond the scope of this study.

1.2 Assessment steps

The assessment has involved the following four steps:

1) Description and quantification of current bioenergy feedstock production (baseline) and projections of likely development of bioenergy feedstock production in the EU to 2030 and 2050.

2) Assessment of the sensitivity of EU protected habitats and species potentially exposed to the four bioenergy feedstock types and risk factors affecting this, based on a review of scientific literature and an analysis of the pressures on each EU protected habitat and species reported by Member States under the Birds and Habitats Directive. This was carried for a selection of EU protected habitats and species that are most likely to be exposed to bioenergy feedstock production, i.e. those that occur in cropland, grassland, forests, heath, and scrub. Aquatic habitats and species were not included.

3) Semi-quantitative assessment of the vulnerability of each EU protected habitat and species potentially exposed to the four bioenergy feedstock types, based on the estimated exposure of each to each feedstock type (from step 1) and their sensitivity to it (from step 3). The assessments are in relation to:
    a. the potential impacts assuming only basic good practice mitigation measures are in place; and

b. the potential impacts assuming all existing mitigation measures (e.g. protected area designations) are implemented as intended.

2 Description and quantification of current bioenergy feedstock production and projections of bioenergy feedstock production in the EU to 2030 and 2050

This section presents the analysis that attempts to define the current use of EU produced biomass feedstocks for energy, and projections of use in 2030 and 2050. Given the focus of this study on land use and land management in the EU and consequent impacts for habitats and species protected under the Habitats and Birds Directives, the emphasis is on domestic production of feedstocks, while it is noted here that feedstocks are also supplied from non-domestic sources.

The following are presented in this analysis

- A synthesis of estimates of bioenergy consumption in EU to 2020, 2030 and 2050 generically in order to determine likely future trends;
- An assessment of existing and anticipated future demand based on the different feedstock types;
- An extrapolation of the data on estimated feedstock use (current and in 2030) to support the assessment of the vulnerability of EU protected habitats and species (in Section 4).

Where-ever possible, official Eurostat data have been used as a basis for this analysis. Where these are lacking or limited, other sources have been used to compliment and support the pan European data sets. Additional sources have been used for forestry statistics and specialist energy crops, and to make the link between bioenergy production and biomass feedstock use.

2.1 EU Policy Context – Driving Bioenergy Demand in the EU to 2020, 2030 and 2050

Since the adoption of Directive 2009/28/EC[4] (the RED), and associated binding targets for total renewable energy use and use of energy from renewables in the transport sector[5], EU policy has been a key driver for the promotion of biomass use for energy within the European Union. Whilst the EU policy sets no target for bioenergy use per se, demand results from the wider promotion of renewables combined with the relative flexibility and transformational adaptability of biomass fuels. In 2015, 64% of total primary energy production of renewable energy in the EU-28 was generated from biomass (Eurostat[6]). Wood and solid biomass, liquid biofuels, biogas and renewable wastes ie all biomass accounted for 8.6% of total EU 28 energy consumption in 2017, representing approximately 62% of total renewable energy consumption in the EU (Eurostat[7]). Analysis of Member States use of biomass for energy

[4] Adopted in parallel to greenhouse gas intensity targets for transport fuels under the Fuel Quality Directive (2009/30/EC)

[5] 20% of energy use be from renewable sources by 2020 (which is differentiated into nationally binding targets per MS) and that at least 10% of transport fuels come from renewable sources by 2020 to be delivered by all Member States

[6] http://ec.europa.eu/eurostat/web/environmental-data-centre-on-natural-resources/natural-resources/energy-resources/energy-from-biomass

[7] Share of renewables in gross inland energy consumption, 2017 - https://ec.europa.eu/eurostat/statistics-explained/index.php?title=File:Share_of_renewables_in_gross_inland_energy_consumption,_2017_(%25).png

(based on National Renewable Energy Action Plans (NREAPs) designed to demonstrate compliance with the RED targets to 2020) has shown an anticipated doubling of biomass use for energy, from 5.4% of final energy consumption in 2005 to almost 12% (124Mtoe) in 2020 driven by the RED.

Up to 2020, the RED sets out sustainability criteria specifying rules to be complied with if transport biofuels or bioliquids used for heat or power are to be counted towards the targets set in RED or supported by national policies and funds. These criteria include:

- minimum GHG emission savings to be achieved; and
- rules intended to avoid the sourcing of feedstock material for the production of biofuels and bioliquids from land considered of high biodiversity value or of high carbon stock in January 2008.

The RED land-based sustainability criteria are intended to avoid direct land use change linked to increased demand for biofuel or bioliquid feedstocks associated with the EU targets. They prohibit sourcing of material from areas including primary forest and other wooded land, designated areas for nature conservation, highly biodiverse grasslands, wetlands, continuously forested areas, and peatland. It should be noted that no EU level sustainability criteria exist for biomass used in solid or gaseous form for energy production up to 2020, although some Member States have developed their own national rules.

Directive 2009/28/EC was subsequently amended by Directive 2015/1513 with the intention of limiting indirect land use change impacts associated with increased demand for biofuel and bioliquid feedstocks. While Directive 2009/28/EC set out sustainability criteria to limit direct land use change, indirect land use change was considered to be occurring at scale i.e. where exiting production was being displaced onto other land to make way for biofuel feedstocks. Directive 2015/1513 increased GHG emission saving requirements and applied a limit to the contribution biofuels made from crop-based feedstocks could make to the renewable transport fuel target (capped at 7 per cent of transport fuels).

Post 2020, the recast of the Renewable Energy Directive (EU(2018)2001) (known as REDII) sets an EU level binding target of 32% of energy from renewable sources by 2030, and a transport fuel mandate requiring 14% of transport fuels from renewable sources by 2030.

Post 2020, the proportional contribution from biofuels and bioliquids, (as well as of biomass fuels consumed in transport), produced from food and feed crops remains capped at no more than one percentage point higher than the share of such fuels in the final consumption of energy in the road and rail transport sectors in 2020 in that Member State. The Member State can have a maximum of 7% of final consumption of energy in the road and rail transport sectors. Moreover, contributions of biofuels, bioliquids and biomass fuels produced from food and feed crops classed as high indirect land- use change-risk i.e. for which a significant expansion of the production area into land with high-carbon stock is observed, shall not exceed the level of consumption of such fuels in that Member State in 2019. From 31 December 2023 until 31 December 2030 at the latest, that limit shall gradually decrease to 0%. Food and feed crops could only then be used as feedstocks for biofuels, bioliquids or biomass fuels if certified to be low indirect land-use change risk.

The REDII also redefines sustainably standards to be met by biomass used for energy. Post 2030 all agricultural and forest biomass used as biofuels, bioliquids and biomass fuels will be required to meet sustainability criteria, with exemptions applied to non-agricultural and non-forestry-based residues and wastes.

2.2 Estimating the scale of future biomass demand and potential impacts

The sustainability criteria to 2020 and 2030 set out rules that operators have to comply with in order to use biomass for energy, but they do not define the extent to which biomass will be utilised to meet the given targets. This is because bioenergy's contribution to the 2020 and 2030 renewable energy targets and transport fuel target and mandate is neither limited nor required by REDII. It depends on interactions between demand and cost of other renewable energy sources, wider energy infrastructure and rural development investment decisions and competition in terms of demand for agricultural and forest biomass. Hence a number of studies have been completed in order to model the likely demand for biomass for energy, the consequences of that demand in terms of land use change and associated biodiversity impacts. This review synthesises key findings from these studies (see Annex 2 for further details).

It is important to note that the review found no study that combined a renewable energy ambition of 32% renewable energy use by 2030 with land use change assessments. This is because key studies were drafted to support the development of policy proposals aimed at delivering a lower 27% renewable energy target[8]. Therefore, the analysis from earlier studies has been interpreted in a qualitative way to provide some context in terms of likely land use consequences that may have impacts on EU protected habitats and species.

The only study to estimate biomass demand for energy at the scale demanded by the RED II is analysis by IRENA (IRENA and European Commission, 2018). This study noted that biomass demand would grow substantially to 9.6EJ by 2030 to deliver 27% RES; under a scenario to deliver up to 34% RES it would continue to grow to deliver 12.2EJ (roughly a twofold increase between 2010 and 2030). In terms of overall renewable energy demand, biomass from energy expands in terms of total volume used; however, it declines as a proportion of total RES (as the pool of other RES technologies expands faster). This expansion in biomass use volume is focused on transport biofuels;[9] within district heating; and within heating and cooling for industry.

It should be noted that the volumes of biomass to be used for energy under the IRENA modelling exercise to deliver a 27% target are relatively consistent with the scale of biomass energy considered to be needed in other modelling exercises based on estimates produced by the PRIMES and Green X models. This implies that material needed to produce 12.2EJ will be additional to the biomass use estimates based on 27% RES consumption, implying additional land use consequences on top of those predicted in the Green X (Biosustain study)

[8] The ambition of REDII renewable energy targets were increased during negotiations to approve the Directive in the European Parliament and Council from 27% to 32%. Original Commission proposals for the Directive also did not include a renewable transport fuel target or mandate post 2020.

[9] It should be noted that the analysis predates the RED II rules potentially restricting what can be used both in terms of food and feed crops and high ILUC risk fuels and promotion of advanced fuels

and Globiom (RECEBIO studies) modelling exercises discussed below (the latter based on PRIMES estimates).

2.2.1 Land use implications associated with future bioenergy demand

It should be noted that all the studies that analysed land use consequences associated with biomass for energy excluded the use of feedstock material originating from protected areas, including Natura 2000 sites and nationally designated sites. The studies, therefore, assume that habitats and species within Natura 2000 sites are fully protected against negative impacts of bioenergy. However, evidence from the literature review on the sensitivity of EU protected habitats and species (section 3) reveals that in practice this is not the case (as impacts arise from e.g. grassland conversion and agricultural improvement, afforestation and increased forest exploitation and management intensity). This assumption is therefore not made in the initial stage of the vulnerability assessment carried out in section 4.

The Biosustain study (PwC et al, 2017) models the whole range of biomass use for energy and provides some insights into land impacts based on the GreenX model. It should be noted that when considering land use and land use change to deliver biomass feedstocks for energy GreenX excluded protected areas and also only allowed the production of energy crops on so called 'surplus land' (although the study notes this is unlikely to be the case in reality). Surplus land is defined as 'not needed for other purposes including food production'. The study estimated that land used for energy crops will increase to 8 Mha in 2020 compared to 6 Mha in 2012 driven by increased demand for food-based energy crops; however, between 2020 and 2030 land required for food and feed-based crops declines whilst land cultivated with lignocellulosic energy crops is projected to increase 7 fold from 0.14 Mha in 2013 to 1 Mha in 2030.

The Biosustain study concludes that there will be limited impact on biodiversity given the protection of designated areas, decline predicted in the use of food and feed-based fuels and the potential benefits associated with increase in lignocellulosic crops. As noted above, protected area designations do not in fact lead to full protection from impacts arising from bioenergy production. The study does include the caveat that impacts will depend on the location of the newly established cropping systems and the pre-existing status of that land. It should be noted that the study's focus on 'surplus land' in terms of biodiversity impact is not specifically examined; however, this is anticipated to focus energy crop production on semi-natural land or unmanaged abandoned land, outside protected areas. This has implications for the intensity of agricultural land use. While crops may be different in nature to food and feed-based crops, this does not necessarily provide for the necessary mix of habitats or management conditions of the energy crops to support sensitive species.

The emphasis on conversion of semi natural or abandoned agricultural land for energy crops is echoed by results from three studies based on the Globiom model. These look at biomass for heat and power and separately land use change (LUC) associated with biofuels. All three studies conclude that SRC expansion comes at the expense primarily of other natural land (i.e. not forests or protected habitats but other semi natural habitats) and abandoned agricultural land.

In terms of forest-based biomass, two studies using Globiom modelling, RECEBIO I (Forsell et al, 2016a) and RECEBIO II (Forsell et al, 2016b), looked at the impacts of using solid biomass in particular in the heat and electricity sectors. The RECEBIO I study noted a significant shift from unmanaged forest into management to deliver 27% renewable energy to 2030. The RECEBIO II did not note this trend but noted an increase in the used forest area and SRC at the expense of other natural land categories. However, in simulations where restrictions were applied to the sourcing of biomass material (including limiting SRC expansion) an impact on unused forest was noted. Impacts including loss of other natural land and conversion of unused forest also increased when scenarios simulated rising demand both in Europe and increased biomass demand in the rest of the world for biomass for energy (reducing import potential into Europe).

In summary, the increased demand envisaged within the IRENA study for biomass to deliver a higher renewable energy target to 2030 is likely to be sourced primarily from woody biomass or ligno-cellulosic biomass. This is due to increasing restrictions over the 2020 to 2030 period on the use of food and feed-based biofuels and biomass for energy: and the reduction in high ILUC biofuels and bioliquids over the same period. Because of the increased scale it is anticipated that there will be impacts on forest systems (i.e. conversion of unmanaged to managed forest) and that SRC and forest land will expand within the EU. The extent to which impacts fall within the EU will, however, depend strongly on the profile of imports from third countries, which are also increasing.

In terms of land use impacts and their implications for biodiversity, all studies predict that there will be a loss of semi natural land and reconversion of abandoned agricultural land primarily to ligno-cellulosic crops and short to medium term tree rotations. Moreover, based on the anticipated increase in demand for biomass to meet the 32% target, intensification of forest management would also be anticipated, including the conversion of unmanaged forest to management. It should also be noted that, within RECEBIO I analysis, increasing wood volumes were seen before additional land was converted to managed forest. This implies more intensive management of existing forests to deliver biomass demand.

2.2.2 Looking to 2050

Few of the studies looking at biomass management look to the 2050 time horizon. The RECEBIO I and II analysis do look to 2050, where they see a continued, rising demand for wood-based products up to 2050. This is in part driven by wider societal trends i.e. population expansion, assumptions regarding wood use more widely in the economy. However, RECEBIO I notes that post 2030, based on the assumption of an 80% GHG emission reduction target to 2050 for Europe, bioenergy is anticipated to overtake material use as the primary driver of wood extraction. However, demand for biomass for material use from other sectors is anticipated to remain, given the growing policy push towards a bioeconomy.

Assuming bioenergy's role in future energy systems is not restricted and that the EU does not rely on increased imports for its bioenergy feedstocks, it would be anticipated that intensity of forest management would continue to increase between 2030 and 2050 to meet demand. Up to 2050 one might also see new afforested areas becoming of increasing importance, which may offer opportunities for biodiversity depending on management and the land use replaced.

2.3 Assessing existing and anticipated future demand for biomass feedstocks

It should be noted that the scope of this work focuses on EU production of bioenergy feedstocks and their potential consequences for EU protected habitats and species. However, the ahead of the discussion below on feedstocks it is useful to note the level of imports associated with different biomass streams to inform the interpretation below. For soild biofuels, in 2016, 8,026 ktoe were importated out of a total market of 102,151 ktoe. The import of solid biomass has slowly increased from 5.8% in 2011 to 7.8% in 2016. In relation to biogas production, most of these materials are wastes with a high moisture content and costly to transport over a long distance. The energy crops that have been used (often as a co-digestate) in anaerobic digestion are generally produced on or near the farm where the biogas production takes place, since it is not economically attractive to procure such crops from further away (Ecofys and SEI, 2019).

About 64% of the biodiesel consumed in the EU in 2016 came from EU feedstock, mainly from rapeseed (38%), with small amounts from used cooking oil (13%), animal fat (8%) and tall oil (2.5%). The main third countries of origin are Indonesia (14%) and Malaysia (7%) relating to palm oil imports primarily. In 2016, at least 65% of the bioethanol consumed in the EU was from EU feedstocks. This mainly concerns wheat (25%), corn (22%) and sugar beet (17%). The most significant third country contribution was of corn from Ukraine (9%) (Ecofys and SEI, 2019).

2.3.1 Bioenergy from conventional food crops

The primary uses of conventional crops for bioenergy are for biodiesel, bioethanol, and biogas. However, the exact proportions of crops used for bioenergy are not recorded in EU statistics.
It should be noted that while grass can be used as an energy feedstock this is not analysed in detail here. This is because grass use to date is normally integrated within feed cycles or wider management cycles. It is, therefore, not possible to definitively distinguish activities and use patterns.

Figure 2-1 presents analysis by AEBIOM (AEBIOM, 2017) as a potential basis for comparison. It should be noted, however, that the data are derived from modelled scenarios rather than real world measurement and the Eurostat data do not provide a clear basis for defining these proportions. Other sources of differentiation into end uses have not been identified, hence the information in the figure is provided as a best estimate. It shows that whilst only a small proportion of the EU's wheat and maize crops are used for bioenergy, almost half of the rapeseed production goes into bioenergy. In the following, we put this in the context of overall agricultural land use in the EU (see Box 2.1).

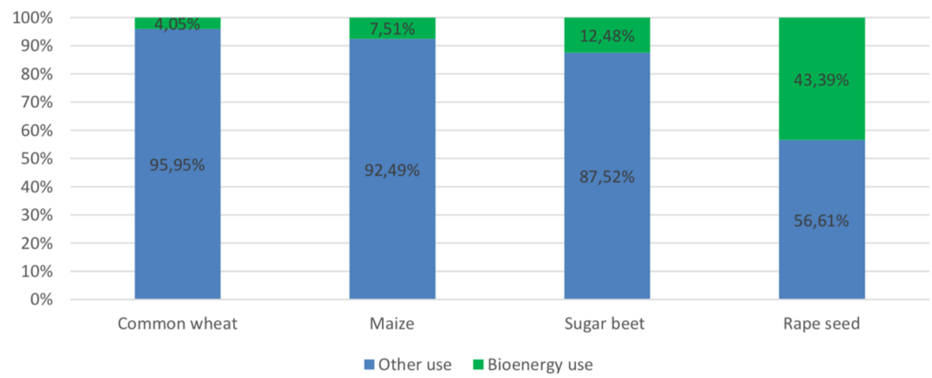
Box 2.1 EU agricultural land uses

According to Eurostat data[10] [11] the EU-28 total Utilised Agricultural Area (UAA) covered nearly 179 million ha in 2015. Of this, 107 million ha (59.8%) consisted of arable land and 59 million ha was occupied by permanent grassland (33.0%), totalling 166 million ha. The main crops grown on arable land in the EU-28 in 2015 were cereals (including rice), which occupied around 57 million ha (53% of arable land). Cereals together with plants harvested green (11.9% of the UAA), industrial crops (7.0%[12]) and fallow land (3.9%) covered 92.3% of the total arable land. The remaining area was dedicated to the growing of root crops (1.7% of the UAA), dry pulses (1.2%), vegetables (1.1%), and other arable crops (0.6%). In the Nordic Member States, almost the entire UAA was taken up by arable land: 98.6% in Finland, 89.2% in Denmark and 85.0% in Sweden. Conversely, the proportion of arable land in total utilised agricultural area in 2015 was below 50% in six Member States: Austria (49.5%), Luxembourg (47.9%), Slovenia (35.9%), the United Kingdom (33.6%), Portugal (30.2%) and Ireland (10.2%). These countries also had the highest shares (close to 50% of the total UAA or more) of permanent grassland, explained by pedologic and climatic factors and high numbers of grazing animals[13].

It should be noted that while grass can be used as an energy feedstock this is not analysed in detail here. This is because grass use to date is normally integrated within feed cycles or wider management cycles. It is, therefore, not possible to definitively distinguish activities and use patterns.

Figure 2-1 Biomass feedstock in 2006 and in the NREAP estimations for 2020

Source: (AEBIOM, 2017)



Biodiesel production – a focus on oilseed feedstocks

Analysis of biodiesel production suggests a number of trends in terms of crop-based feedstocks produced in Europe on agricultural land (Figure 2-2). The use of rapeseed-based

[10] https://ec.europa.eu/eurostat/web/agriculture/data/database

[11] A note on data on crop production in Europe: the EUROSTAT statistics hand book notes that information on the use of crops for energy are included within the wider cropping categories i.e. there is no differentiation based on the end use: Maize for energy purposes i.e. biogas primarily G3000 Green maize; Rape for energy purposes in class i1110 Rape and turnip rape seeds; Sugar beet for the production of energy R2000. Therefore, EU data for conventional crops need to be combined to provide a sensible estimate of use for bioenergy.

[12] To note that dedicated energy crops are included within the industrial crops category, however, it should be noted that the energy crop category only covers crops not considered to be covered by any other categories i.e. green maize, which might be used for biofuels will be considered under the G3000 green maize category, rape for energy purposes under the i1110 rape and turnip rape seed classification

[13] EUROSTAT Main Annual Crops Statistics – extracted key messages 1 Jan 2017; update due Feb 2019

oils has declined marginally between 2010 and 2015, however it has fluctuated over this period. This is most likely due to comparative demand and prices on the commodity markets. Use of soybean-based oils has also declined since 2010. However, the use of sunflower oil for biodiesel has doubled from 140,000 tonnes to 330,000 tonnes over the same period.

Figure 2-2 Evolution of the feedstocks for EU28 biodiesel production (in 1,000 tonnes)

Source - (AEBIOM, 2017)

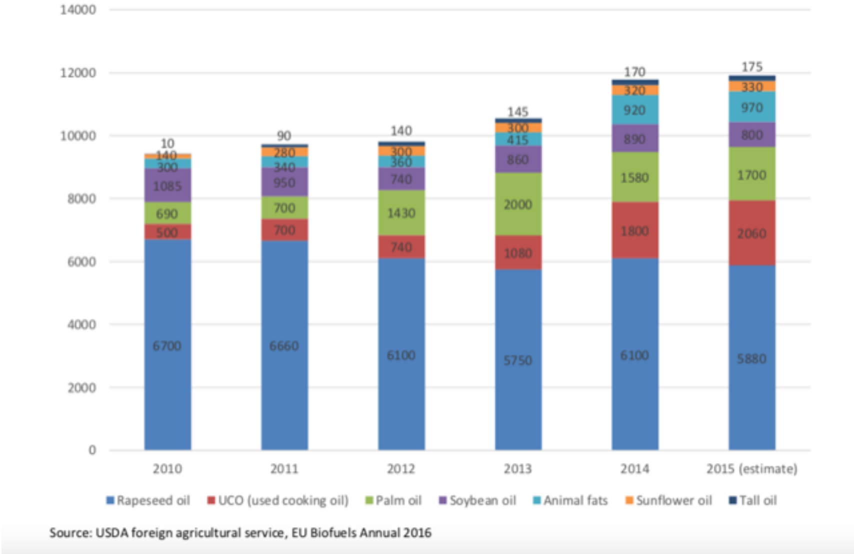


Figure 2-3 Production of oil seeds in terms of production and area in 2015

Source: taken from EUROSTAT Main Annual Crop Statistics, 2017

	Total Rape, turnip rape, sunflower seeds and soya (¹)		Rape & turnip rape seed		Sunflower seed		Soya	
	Harvested production (1 000 tonnes)	Cultivation area (1 000 ha)	Harvested production (1 000 tonnes)	Cultivation area (1 000 ha)	Harvested production (1 000 tonnes)	Cultivation area (1 000 ha)	Harvested production (1 000 tonnes)	Cultivation area (1 000 ha)
EU-28	31 913.5	11 555.1	21 701.0	6 465.3	7 906.4	4 196.9	2 440.1	892.9
Belgium	48.3	11.3	48.3	11.3	0.0	0.0	0.0	0.0
Bulgaria	2 174.9	1 015.7	422.1	170.4	1 699.2	810.8	40.3	34.5
Czech Republic	1 308.1	393.9	1 256.2	366.2	31.6	15.5	20.2	12.3
Denmark	826.0	193.0	826.0	193.0	0.0	0.0	0.0	0.0
Germany	5 086.1	1 315.9	5 016.8	1 285.5	35.3	18.4	:	:
Estonia	196.3	70.8	196.3	70.8	0.0	0.0	0.0	0.0
Ireland	39.9	8.9	39.9	8.9	0.0	0.0	0.0	0.0
Greece	243.5	110.9	3.2	1.7	236.0	107.2	4.4	2.0
Spain	922.5	811.2	149.2	71.0	769.2	738.9	4.1	1.3
France	6 827.1	2 238.8	5 307.2	1 498.6	1 185.8	618.2	334.2	122.0
Croatia	347.3	145.3	56.8	22.0	94.1	34.5	196.4	88.9
Italy	1 393.1	435.7	28.1	12.2	248.0	114.5	1 117.0	309.0
Cyprus	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Latvia	293.2	88.0	293.2	88.0	0.0	0.0	0.0	0.0
Lithuania	513.9	166.2	512.2	163.5	0.0	0.0	1.8	2.6
Luxembourg	13.8	4.0	13.8	4.0	0.0	0.0	0.0	0.0
Hungary	2 293.3	904.2	590.4	220.6	1 557.0	611.6	145.9	72.0
Malta	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Netherlands	8.9	2.3	8.9	2.3	0.0	0.0	0.0	0.0
Austria	286.0	113.5	111.8	37.5	38.1	19.1	136.2	56.9
Poland	2 711.8	954.6	2 700.8	947.1	2.2	1.3	8.8	6.2
Portugal	24.7	19.9	0.0	0.0	24.7	19.9	0.0	0.0
Romania	2 967.3	1 507.5	919.5	367.9	1 785.8	1 011.5	262.0	128.1
Slovenia	8.7	3.6	3.6	1.6	0.6	0.2	4.7	1.7
Slovakia	557.0	238.1	320.0	119.3	174.3	75.4	62.1	43.4
Finland	85.3	55.3	85.3	55.3	0.0	0.0	0.0	0.0
Sweden	359.3	94.5	359.3	94.5	0.0	0.0	0.0	0.0
United Kingdom	2 571.0	652.0	2 542.0	652.0	0.0	0.0	0.0	0.0
Iceland	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Norway	:	3.5	10.4	3.5	:	0.0	:	0.0
Switzerland	100.9	29.7	87.0	23.4	9.8	4.6	4.1	1.7
Montenegro	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
FYR of Macedonia	12.6	7.2	4.1	1.6	8.5	5.5	0.0	0.0
Albania	0.7		0.0	0.0	2.0	0.7	0.5	0.2
Serbia	924.9	363.3	33.4	12.2	437.1	166.2	454.4	184.8
Turkey	1 962.0	757.0	120.0	35.0	1 638.0	685.0	150.0	37.0
Bosnia and Herzegovina	:	:	2.2	1.1	0.5	0.5	10.4	7.0
Kosovo (*)	:	:	:	:	0.6	0.2	:	:

(:) not available

(¹) The national production figures are reported with national humidity degrees, which vary between 7.2 % and 14 %. The EU-aggregate is reported in 9 % standard humidity for other oilseeds and 14 % for soya. This explains the difference in the sum of all EU Member States and the EU-28 total.

(*) This designation is without prejudice to positions on status, and is in line with UNSCR 1244 and the ICJ Opinion on the Kosovo Declaration of Independence.

The main oilseed crops (rape and turnip rape, sunflower seed and soybeans) were grown on 11.6 million ha across the EU Member States in 2015 (10.8% of the total arable land) [14] (Figure 2-3). Concerning rape and turnip rape, out of a total of 21.7 million tonnes harvested in 2015 (68% of the total oilseed production), France was the largest producer with 24.5% of the total production. Other important producers were Germany (23.1% share), Poland (12.4%) and the United Kingdom (11.7%). Sunflower seed production is concentrated in Eastern and Southern Europe. Romania was the largest producer, with 22.6% of the sunflower production in 2015, followed closely by Bulgaria (with 21.5%), Hungary (19.7%) and France (15.0%).

EU harvested production of the main oilseed crops has grown considerably in recent years, namely 31.7% from 2007 to 2015 (Figure 2-4). The increase in production was always higher than the increase of the cultivation area, implying an intensification of production. Specifically, production of:

- Soya in Europe increased by 183.2% from 2007 to 2015, and the 2015 harvest more than doubled if compared to the 2007–14 average (108.9%). Italy, France, Croatia and Hungary were among the countries with the largest increases both in harvested tonnes and in cultivated hectares of soybean;
- Sunflower seed increased by 62.2% between 2007 and 2015, reaching 7.9 million tonnes in 2015. The corresponding cultivation area of sunflower seed also increased by 19.8% from 2007 to 2015 and was 3.1% higher than the 2007–14 average.
- Rape and turnip rape seed presented a more moderate growth during the same time span with an increase of 17.2% (3.2 million tonnes) of the harvested production and a 5.4% increase of the cultivation area.

The statistics on feedstock use for biodiesel production suggest that significant quantities of rapeseed oil are used i.e. it is the primary source of biodiesel in Europe. However, the proportion is not significantly increasing year on year. Soy too appears to be on a downward trend in terms of biodiesel production. The only upward trend is in sunflower oil. As upward trends in EU production were noted for all these oilseeds, there is, therefore, a question over whether biofuel demand is the key driver for rape and soy increases in production in Europe.

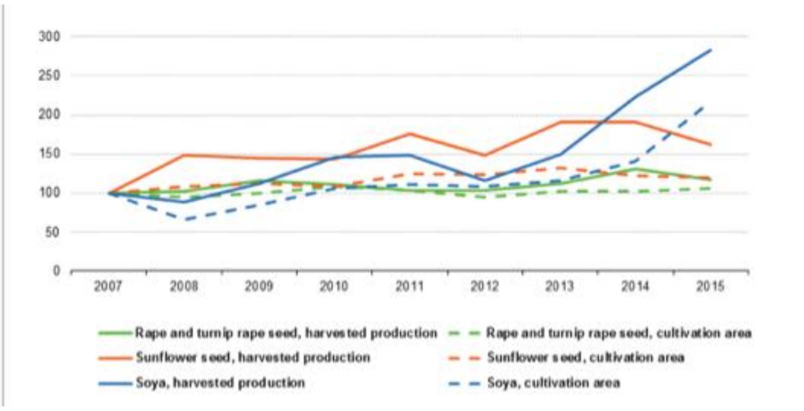
In terms of sunflower oil, it is fair to assume that biodiesel production is contributing to expanded production in Europe. However, it should be noted that there are two trends noted in the summaries i.e. both expanded area, but also increasing yield per unit area. This suggests an increased intensity of production. In Romania, the surface area of rapeseed increased sharply to 2010, partly driven by use for biofuel production, after which rape declined whilst the sunflower production area increased (Vasile et al, 2016).

It is important to note that the statistics do not provide sufficient detail to understand if EU production of feedstocks is increasing compared to imports e.g. within the soy category.

[14] Oil Seed Production in Europe (data extracted from Eurostat – Main Annual Crop Statistics Jan 2017 and Agriculture, Forestry and Fisheries Statistics 2017 edition)

Figure 2-4 Evolution of harvested production and cultivated area of rape and turnip seed, sunflower seed and soya, EU 28, 2007-15

Source of data: Eurostat apro_acs_a. 2007 =100



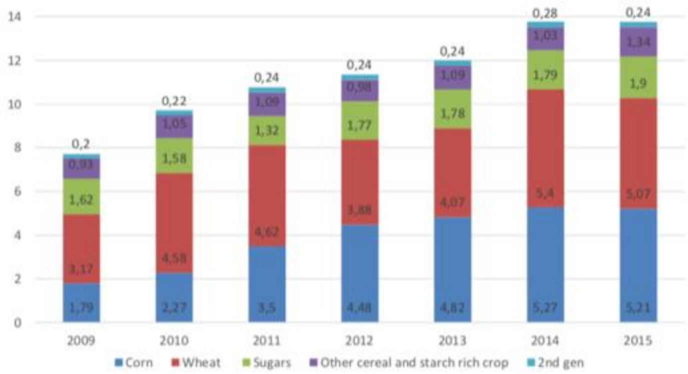
Bioethanol production – a focus on cereal production

The bioethanol statistics for Europe suggest that corn-based ethanol has expanded to the greatest degree in terms of feedstocks used to produce bioethanol in Europe (Figure 2-5). The amount of wheat used as a feedstock has also increased significantly, with a lesser expansion in sugar-based bioethanol.

It should be noted that specifically in the case of corn-based ethanol and sugar-based bioethanol, the primary sources tend to be associated with cultivation of feedstocks in third countries.

Figure 2-5 Evolution of feedstocks for EU 28 bioethanol production (in 1000 tonnes)

Source: (AEBIOM, 2017)



Source: ePURE, European renewable ethanol, aggregate and audited data of ePURE members

Cereal production[15] has fluctuated considerably since 2007 (Figure 2-6). Due to comparatively high cereal prices in 2007 caused by unbalanced supply and demand and unfavourable weather conditions leading to a production decrease in 2009. The downward trend continued in 2010. Despite the production decreases of 2009, 2010, 2012 and 2015 the total level of cereal production in the EU-28 stood nevertheless 20.3% higher in 2015 than in 2007 (an increase of 53.5 million tonnes).

In contrast to production, the harvested area of cereals in the EU-28 remained relatively stable between 2007 and 2015 – never fluctuating by more than 6% (Figure 2-7). When comparing the 2015 values with the 2007–14 average, the discrepancy between the increase in harvested production (+ 6.3%) and the area cultivated with cereals (– 1.6%) suggests a significant improvement in yield.

Cereal production has fluctuated significantly, not following a specific trend. However, there is a suggestion of increasing improvements in yield that may suggest an intensification of production activities. To note that while there is an overall decline in areas of cereal production, the statistics suggest that the areas being cultivated for cereals are shifting with increases in cultivated area being seen in Baltic Member States, Belgium, and Bulgaria[16].

Sugarbeet is also a potential source of domestically produced bioethanol. Given the EU production quotas and increasing yields in the EU-28, the sugar beet area was reduced by 28.4% between 2007 and 2015 (EUROSTAT statistics). The situation differed considerably in the individual Member States. In some Member States sugar beet production declined significantly in 2015 compared to the 2007–14 average: Portugal (– 89.7%), Bulgaria (– 73.7%), Greece (– 64.8%), Sweden (– 48.1%), Croatia (– 38.5%) and Italy (– 37.3%). In other Member States the sugar beet production volumes increased, eg in Romania (+ 20.3%), Slovakia (+ 18.3%) and Denmark (+ 6.2%).

[15] Only cereal production is considered here, while the EU does produce sugar beet. Analysis of the trend data suggests that recent trends in production appear have been primarily linked to production quotas and change in subsidy systems. Note again increasing yields to area of production are significant. Moreover, there is no overarching trend to define and the statistics suggest that there are not major changes in wheat or grain maize to suggest that EU production is shifting to take account of bioethanol production – see annex for details.

[16] In the Baltic Member States as well as in Belgium and Bulgaria there was a significant increase in the cereals cultivation area compared to the 2007–14 average that ranged from 9.7% in Belgium to 21.2% in Latvia.

Figure 2-6 Volumes and area of cereal production in Europe in 2015 by Member State

Source: Main Annual Crop Statistics, 2017

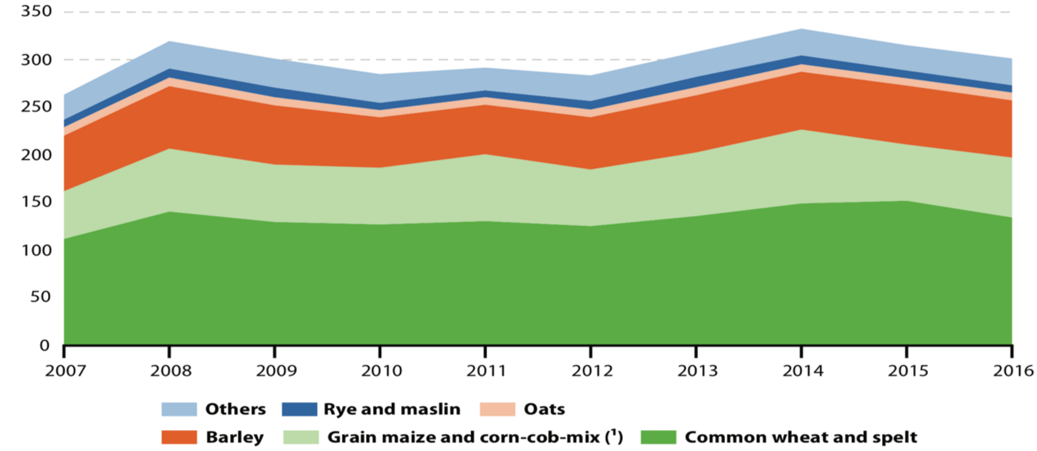
	Harvested production (1)					Cultivation area of total cereals
	Common wheat and spelt	Barley	Grain maize and corn-cob-mix	Rice	Total cereals (2)	
	(1 000 tonnes)					(1 000 ha)
EU-28	152 067.3	61 899.2	58 901.3	2 943.5	316 768.0	57 394.7
Belgium	2 076.3	434.1	693.0	0.0	3 282.5	341.6
Bulgaria	4 979.6	714.8	2 696.9	67.7	8 728.2	1 835.8
Czech Republic	5 274.3	1 991.4	442.7	0.0	8 183.5	1 389.8
Denmark	5 029.0	3 856.0	53.0	0.0	10 023.0	1 453.0
Germany	26 462.3	11 629.9	3 973.0	0.0	48 917.7	6 529.2
Estonia	812.6	556.6	0.0	0.0	1 535.3	350.4
Ireland	696.6	1 739.2	0.0	0.0	2 633.6	291.6
Greece	353.0	353.9	1 542.3	251.2	3 437.1	963.1
Spain	5 437.7	6 705.1	4 564.4	847.0	20 141.0	6 195.9
France	40 910.3	13 027.6	13 738.2	80.9	72 633.2	9 575.6
Croatia	753.0	193.5	1 709.2	0.0	2 796.8	489.7
Italy	2 996.1	930.5	7 069.7	1 466.0	17 553.1	3 011.2
Cyprus	0.0	52.2	0.0	0.0	88.1	32.9
Latvia	2 250.1	385.1	0.0	0.0	3 021.5	669.5
Lithuania	4 380.3	811.5	56.3	0.0	6 066.7	1 329.1
Luxembourg	91.1	44.4	0.9	0.0	176.5	29.3
Hungary	5 237.6	1 408.6	6 632.8	9.4	14 145.2	2 697.7
Malta	0.0	0.0	0.0	0.0	0.0	0.0
Netherlands	1 287.8	226.7	171.0	0.0	1 706.5	195.6
Austria	1 637.3	840.4	1 637.9	0.0	4 843.8	780.7
Poland	10 957.8	2 960.7	3 156.2	0.0	28 002.7	7 511.8
Portugal	74.5	44.4	827.5	184.9	1 241.3	270.1
Romania	7 954.5	1 623.2	8 984.7	49.8	19 286.2	5 466.5
Slovenia (3)	157.1	93.2	338.7	0.0	624.1	99.0
Slovakia	1 968.8	668.7	929.2	0.0	3 805.7	749.2
Finland	992.1	1 569.0	0.0	0.0	3 682.8	1 017.3
Sweden	3 300.4	1 672.3	6.4	0.0	6 168.8	1 019.3
United Kingdom	16 444.0	7 370.0	22.0	0.0	24 735.0	3 100.0
Iceland	0.0	4.6	0.0	0.0	0.0	2.0
Norway	495.2	510.7	:	:	1 357.9	282.7
Switzerland	0.1	197.8	95.4	0.0	891.4	143.9
Montenegro	2.1	1.0	2.7	0.0	7.0	2.3
FYR of Macedonia	201.2	101.7	137.0	30.5	483.8	159.6
Albania	275.0	7.0	380.0	0.0	695.5	142.6
Serbia	2 428.2	362.2	5 454.8	:	8 437.0	1 759.5
Turkey	18 505.0	8 000.0	6 400.0	552.0	38 637.0	11 713.0
Bosnia and Herzegovina	213.0	63.2	785.8	0.0	1 137.6	301.5
Kosovo (*)	304.4	3.1	131.5	:	443.6	134.9

Note: the "cultivation area" corresponds to the area that was cultivated/ harvested during the crop year, it may differ from the "main area" which corresponds to the area of the land parcel, the land use linked to that area is the unique or main crop having occupied the parcel during the crop year.
(:) not available

(1) The national production figures are reported with national humidity degrees, which vary between 10.2 % and 34 %. The EU-aggregate is reported in 14 % standard humidity. This explains the difference in the sum of all EU
(2) Total cereals include also rye, oats, sorghum, buckwheat, canary seed and millet.
(3) 'Common wheat and spelt': definition differs.
(*) This designation is without prejudice to positions on status, and is in line with UNSCR 1244 and the ICJ Opinion on the Kosovo Declaration of Independence.

Figure 2-7 Trends in the production of the main cereal crops in the EU 28, 2007-2016 (million tonnes

Source: Agriculture, Forestry and Farming Statistics, 2017



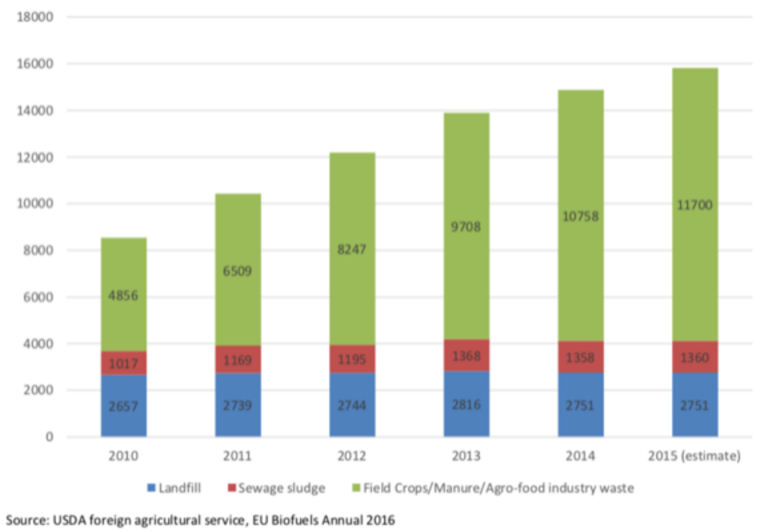
Note: 'Rye and maslin' includes mixture of rye with other winter sown cereals. 'Others' includes rice, triticale and sorghum.
(1) Includes estimates for Denmark 2007-2009 and Sweden 2007-2008.
Source: Eurostat (online data code: apro_acs_a)

Production of biogas in Europe – a focus on crops harvested in the green and green maize production

The biogas market has evolved significantly. Biogas production has expanded, estimated to have almost doubled between 2010 and 2015 and this expansion is primarily based on agricultural inputs[17]. However, all the identified statistics on feedstocks (from European Commission reporting and AEBIOM figures) include only a joint agricultural category (see Figure 2-8). They do not distinguish between the sources i.e. dedicated crop-based biogas, grass-based biogas, crop residue-based material or manure. According to the European Biogas Association, the total number of biogas plants rose from 6,227 to 17,662 installations (+11,435 units) between 2009 (earliest EBA data) and 2016. Growth was particularly strong from 2010 to 2012. Most of that growth derives from the increase in plants running on agricultural substrates: these went from 4,797 units in 2009 to 12,496 installations in 2016 (+7,699 units, 67% of the total increase)[18].

Figure 2-8 Evolution of feedstock for EU 28 biogas production (in ktoe)

Source: (AEBIOM, 2017) statistics



Crops harvested in the green can be used for fodder or biogas production. However, the EU statistics focus on green maize. Green maize was harvested from almost 6.2 million ha in the EU-28 in 2015 (Figure 2-9). The area increased by 0.7 million ha (+13.4%) if compared with 2010 area. The production amounted to 232 million tonnes, nearly 32 million tonnes more than in 2010 (+16%). Germany was the EU-28's leading producer of green maize with more than one third of total production (slightly over 87 million tonnes harvested from about 2.1

[17] According to the European Biogas Association between 2009 (earliest EBA data) and 2016, the total number of biogas plants rose from 6,227 to 17,662 installations (+11,435 units). Most of that growth derives from the increase in plants running on agricultural substrates: these went from 4,797 units in 2009 to 12,496 installations in 2016 (+7,699 units, 67% of the total increase) - http://european-biogas.eu/2017/12/14/eba-statistical-report-2017-published-soon/

[18] http://european-biogas.eu/2017/12/14/eba-statistical-report-2017-published-soon/

million ha in 2015). Germany and four other Member States (France, Poland, Italy and the Czech Republic), together accounted for over three quarters of total EU-28 green maize area. Between 2010 and 2015, significant increases in production of green maize were observed in Estonia (+ 105.2% above the 2010–14 average) and Latvia (+ 54.1% above the 2010–14 average).

The area of green maize is noted to be expanding, but production and expansion are concentrated in specific Member States. This is likely to be a result of demand for feed as well as to meet biogas demand. It should be noted, however, that biogas demand in leading production Member States, and in Estonia and Latvia, has grown over the period from 2010-2015. In Germany, for example, biogas subsidies have driven a rapid increase in the area of maize since 2009, although the rate of increase has slowed since 2013 (FNR, 2018). Maize has replaced summer cereals and sugar beet, temporary and permanent grassland in Germany (KLU, 2013). Although the main driver for the increase in maize is generally the need for silage maize for intensive dairy production and high grain maize prices for animal feed, the demand for biogas feedstocks has driven increases in maize cultivation in the arable area surrounding biogas plants (Laggner et al, 2014).

Figure 2-9 Production and cultivated area for green maize in 2015

Source: Main Annual Crop Statistics, 2017

	Harvested production (¹) (1 000 tonnes)	Humidity degree (%)	Cultivation area (1 000 ha)
EU-28	231 652.9	65.0	6 187.2
Belgium	8 009.3	35.7	173.3
Bulgaria	512.8	65.0	26.6
Czech Republic	7 134.4	83.0	245.0
Denmark	5 605.0	45.0	184.0
Germany	87 218.9	65.0	2 100.4
Estonia	230.8	70.0	8.5
Ireland	562.9	64.6	12.9
Greece	146.9	42.0	13.5
Spain	4 473.6	65.0	107.9
France	17 259.4	0.0	1 475.2
Croatia	1 159.3	65.0	32.6
Italy	16 668.6	58.6	336.9
Cyprus	19.3	:	0.3
Latvia	730.2	70.0	25.4
Lithuania	771.2	60.0	29.3
Luxembourg	178.4	0.0	14.5
Hungary	2 154.4	60.0	90.0
Malta	0.0	:	0.0
Netherlands	8 222.4	66.1	223.9
Austria	3 807.1	68.0	92.0
Poland	19 801.8	80.0	555.2
Portugal	3 152.2	75.0	80.8
Romania	1 239.4	65.0	46.3
Slovenia	1 398.5	65.0	28.7
Slovakia	2 054.3	75.0	89.5
Finland	0.0	0.0	0.0
Sweden	165.9	0.0	15.7
United Kingdom	:	65.0	179.0
Iceland	0.0	:	0.0
Norway	:	:	0.0
Switzerland	704.0	70.0	45.9
Montenegro	0.0	:	0.0
FYR of Macedonia	142.4	:	5.1
Albania	481.0	:	15.3
Serbia	589.2	:	34.1
Turkey	19 920.0	:	423.0
Bosnia and Herzegovina	532.6	65.0	31.9
Kosovo (²)	31.6	65.0	2.3

(:) not available

(¹) Green maize production is reported with national humidity degrees, which vary from 0 % (dry matter) to 83 %. For the EU-aggregate the national production figures are standardized to 65 % humidity. This explains the difference in the sum of all EU Member States and the EU-28 total.

(²) This designation is without prejudice to positions on status, and is in line with UNSCR 1244 and the ICJ Opinion on the Kosovo Declaration of Independence.

It should be noted that analysis for Germany has identified cob maize being used for biogas use as well as green maize. Total area for grain maize in 2015 for the EU 28 was 9,255.56 thousand hectares (Eurostat crop statistics). As noted in
It should be noted that while grass can be used as an energy feedstock this is not analysed in detail here. This is because grass use to date is normally integrated within feed cycles or wider management cycles. It is, therefore, not possible to definitively distinguish activities and use patterns.

Figure 2-1, 7.5% of total maize was estimated to be used for energy in 2015.

Future demand for conventional food crops for bioenergy

The main policy driver is towards a reduction in the use of conventional food crops for biofuel and wider biogas production. Looking to 2030 REDII would apply sustainability requirements to all agricultural biomass used for energy (i.e. including biogas) post 2020. Moreover, building on an existing cap set in Directive (EU) 2015/1513 intended to limit the use of biofuels and bioliquids from food and feed-based crops, the RED essentially limits the use of conventional food crops at 2020 levels to 2030[19]. In addition, the limits on 'high indirect land-use change risk food or feed crop-based biofuels, bioliquids and biomass fuels produced from food and feed crops'[20] may result in further declines in certain European based food and feed based feedstocks[21].

According to European Commission's Agricultural Outlook (2018-2030)[22] 'the cap on biofuels produced from food and feed crops (to be set in 2020) could translate into a further increase in crop-based biofuel consumption in 2019 and 2020. The outcome of the assessment on high-ILUC-risk feedstocks and the results of applying the future methodology on low-ILUC certification of crop-based biofuels will have a significant impact on the types of feedstock and biofuel consumed in the biofuels mix in the EU. Overall, the share of crop-based biofuels, in energy terms, is projected to increase from 4.7 % in 2020 to 5.5 % in 2030, driven by biofuel mandates set by Member States. However, it should be noted that these figures include imports as well as domestic production. When only domestic production is included there is a drop in first generation biofuel production from European based food crops. This sees biodiesel production from food and feed-based crops in Europe remain relatively static and equivalent bioethanol production decline slightly (i.e. around 4%) [23]. It should be noted here

[19] As set out in article 35 of RED II, the use of food and feed based conventional biofuels would be capped at *a maximum of 7%* of gross final consumption in road and rail transport in that Member State/or no more than 1 percentage point higher than their contribution in 2020 (Directive 2015/1513/EU also sets a limit of 7%.

[20] RED II, Article 25

[21] The limits are set in COMMISSION DELEGATED REGULATION (EU) …/... supplementing Directive (EU) 2018/2001 as regards the determination of high indirect land-use change-risk feedstock for which a significant expansion of the production area into land with high carbon stock is observed and the certification of low indirect land-use change-risk biofuels, bioliquids and biomass fuels. C/2019/2055 final

[22] https://ec.europa.eu/info/sites/info/files/food-farming-fisheries/farming/documents/medium-term-outlook-2018-report_en.pdf

[23] It should be noted that the Agricultural Outlook data is used rather than studies completed for the impact assessment of the RED II proposals. This is because the policy baseline has changed significantly as a result of amendments made during the approval of REDII, the Agricultural Outlook 2018 takes this into account

that imports of biofuels and biofuel feedstocks are currently associated with negative impacts on biodiversity outside Europe (Bowyer, 2010a; Meletiou et al, 2019).

2.3.2 Agricultural residues from crops

The total agricultural biomass produced annually in the European Union is estimated at 956 Mt of dry matter, as averaged from 2006 to 2015 (Camia et al, 2018). The fraction of the biomass which is not the primary aim of the production process (e.g. dry biomass from leaves, stems), is referred to as residue production, although sometimes residues may generate farm income (e.g. animal bedding production of bio-energy). Residues are also essential for other uses including ecosystem services such as maintaining soil organic carbon levels in the soil or preventing soil erosion. Production of agricultural residues in Europe is estimated as 442 Mt or 46% of total agricultural biomass (Camia et al, 2018).

Information on agricultural residues is limited, the proportion of residues being produced associated with a given crop, or the volume of residual material is not recorded. The information, for example, about volume or area of cereal straw would not be differentiated from the areas of overall cereal production. Hence estimates of agricultural residues by the JRC (Camia et al, 2018) are based on empirical models[24]. The uncertainties are relatively large indicating the need for future improvements in the models used to estimate agricultural residue production.

Based on analysis by JRC in relation to residue production, it is estimated that: cereals represent 74% of total agricultural residue production (329 Mt/yr), while oil-bearing crops are the second group of importance with 17% (73 Mt/yr). Total residue biomass from agricultural has increased slightly between 1998 and 2015. This is linked to expansion in areas of ‘high residue producing crops’ e.g. oil seeds.

Estimating extraction rates for residues is equally challenging. Some studies have estimated the available potential of agricultural residues. For example, analysis looking at available wastes and residues for advanced biofuels (Searle and Malins, 2013) estimated that around 122 million tonnes of agricultural residues are currently sustainably available; based on the assumption that one-third of residues must remain in the field to maintain soil quality, and one-third must be left for existing uses. Biomass Futures work has identified a straw potential of 127 million tonnes for the EU-27 in 2020 (Elbersen et al, 2012). Analysis by JRC (Camia et al, 2018) has attempted to estimate the share of collected crop residues; i.e. the fraction of

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[24] Residue production is currently deduced from economic production using empirical models that describe, for each individual crop, the relationship between biomass cumulated in plant storage organs and biomass produced in other aboveground organs (e.g. leaves, stems, etc.). Although relationships exist for most crops, estimating residue production solely from economic production is an over-simplification, as genetic factors (varietal differences), agro-climatic conditions, and agro-management practices (e.g. irrigation, fertilizing) influence the relationship. As a result, the uncertainties can be large in most of the model estimates for specific yield intervals (Camia et al, 2018).

crop residues which is collected and enters bioeconomy value chains. This is estimated to range from 25% for cereals to 10% for oil bearing crops and 0% for pulses (see Figure 2-10).

Agricultural residues are often discussed in the context of advanced biofuels. However, data on advanced bioethanol production (Figure 2-5) shows limited adoption in 2015. Agricultural residues can also be used in other forms of biomass for energy including as inputs to pellet production, biogas, or dedicated biomass plants. However, there is no coherent data on the extent of their use to date. Expansion in the use of agricultural residues is seen as a key source of biomass for energy and use is anticipated to expand to 2030 to replace food and feed-based fuels.

Figure 2-10 estimates of current extraction rates of agricultural residues per crop type

Source: Data from JRC (Camia et al, 2018). To note that this is unlikely to encompass significant shares for biofuels, although some elements might be used on a small scale for wood chip, heat energy or biogas.

Crop type	Share of collected crop residues
Cereals	25%
Fruit trees and berry plantations	10%
Vineyards	10%
Cotton fibre	0%
Fibre flax	0%
Hemp	0%
Other fibre crops n.e.c.	0%
Hops	10%
Tobacco	10%
Olive trees	10%
Oil-bearing crops	10%
Pulses	0%
Potatoes	10%
Nuts	10%
Vegetables, melons and strawberries	10%
Plants harvested green	0%
Sugar beet	50%

2.3.3 Energy crops, focusing on novel crops and non-food crops

Miscanthus, SRC and similar energy crops

Eurostat contains data on energy crops; however, they hold only one combined indicator and it is noted to be based on inconsistent reporting. The energy crop statistics were originally used as a basis for monitoring energy crop incentives under the CAP. However, once these were removed, through the removal of set aside, the statistics were not consistently recorded by Member States[25]. Most publications looking at energy crops use AEBIOM data on energy crop patterns, which are based on their own surveys of members (bioenergy producers). The data reported in AEBIOM's 2017 statistical report (AEBIOM, 2017) state that there are in total approximately 51 ha of dedicated energy crops in Europe with just under 15 ha of Miscanthus and just over 37 ha of short rotation coppice (Figure 2-11). These totals are considered to be too low based on estimated of SRC area in 2012 in a limited number of Member States in the SRCplus project (Dimitriou et al, 2014). The project reported 2,457 ha SRC in France, 2,088 ha SRC in the Czech Republic, and between 4000 and 6000 ha SRC in Germany. National data for the UK[26] and France[27] suggests that in 2015 they had 6,905 ha and approximately 5,000 ha of Miscanthus respectively. However, it should be noted that France considered only two thirds of Miscanthus to be currently in use for energy, the other third is used primarily for mulching and animal bedding.

Modelling studies[28] produced for the European Commission anticipate a significant increase in the area of energy crops, both SRC and other perennial lignocellulosic crops. The RECEBIO I analysis (Forsell et al, 2016a) identified an increase in SRC under its BAU scenario from 10,000ha in 2010, to 2,500,000 ha by 2030, to 3,400,000 ha in 2050, and under its EU Emission Reduction Scenario (where ambition and use of bioenergy increases to 2050) 8,900,000 ha in 2050. It should be noted that the 2010 data are model outcomes run from 2000 base year.

Advanced bioethanol is anticipated to increase and expand; and that wood use more generally for energy will also do so, based on predicted levels of bioenergy use to 2020 and 2030. The use of SRC is seen as an alternative to intensification of forest uses or imports of biomass material from third countries[29]. It should be noted that SRC expansion in particular is seen within the modelled results as occurring on 'other natural land' (RECEBIO - Forsell et al, 2016a; Forsell et al, 2016b) or 'surplus land' (Biosustain - PwC et al, 2017). This is land not currently in active use for agricultural or forestry production, and not protected for nature conservation. However this often includes other semi natural land and scrub land potentially of biodiversity importance.

[25] Noted as such in the statistics handbook

[26] https://assets.publishing.service.gov.uk/government/uploads/system/uploads/attachment_data/file/664991/nonfood-statsnotice2016-6dec17b.pdf

[27] https://www.france-miscanthus.org

[28] Recebio I and Recebio II and Biosustain studies all completed for the European Commission in support of the impact assessment of the RED II proposals.

[29] Based on RECEBIO I sensitivity analysis

Figure 2-11 Energy crops in EU 28 (in ha)

Source: (AEBIOM, 2017)

	Miscanthus	Willow (SRC)	Poplar (SRC)	Total Short rotation coppice	Total	Year
EU28	14.788	25.880	12.675	38.829	50.764	
AT	1.128	N.A.	1.500	1.500	2.628	2017
BE	227	N.A.	N.A.	42		2017
BG						
CY						
CZ						
DE		N.A.	5.000	5.000		2017
DK	N.A.	5.500	N.A.	5.500		2015
EE						
EL						
ES						
FI						
FR	3.000	N.A.	N.A.	220		2015
HR		0	0	0	Experimental field only	2017
HU						
IE	700	1.100	0	1.100	1.800	2017
IT						
LT						
LU						
LV						
MT						
NL		N.A.	N.A.	13	245	2016
PL	733	7.728	3.175	10.903	22.539	2013
PT						
RO						
SE		11.552	3.000	14.552	14.552	2016
SI						
SK						
UK	9.000	N.A.	N.A.	N.A.	9.000	2017 (estimate)

Source: Data collected by AEBIOM; AT: ABA;BE: Valbiom & Provinciaal Agrarisch Centrum; DE: Markus hartmann; FR: ADEME; IE: Teagasc; HR: BEECO; NL: Centraal Bureau voor de Statistiek; PL: POLbiom; SE: SVEBIO; UK: Terravesta

Afforestation of agricultural land

Expansion in forest area is noted as a trend by the EEA (EEA, 2015a) and in data for the EU 28 recorded by Forest Europe (Forest Europe, 2015). In 2015, 160,931,000 ha of forest were noted in the EU 28 (Forest Europe, 2015). Between 2010 and 2015 this had risen by 1,695,000 ha (i.e. an annual average of 339,000 ha per year) (Figure 2-12). It should be noted that part of this is active afforestation and part is natural expansion, i.e. for example following abandonment of other land uses. Evidence from Forest Europe[30] suggests high variability by region in active vs natural afforestation, depending on the European region. Only the portion of actively afforested land is considered to be potentially driven by bioenergy demand.

30 Forest Europe 2011 - https://www.foresteurope.org/docs/SoEF/reports/Criterion_1_Forest_Resources_and_their_Contribution_to_Global_Carbon_Cycles.pdf

Figure 2-12 Annual change in forest area by region

Source: (Forest Europe, 2015)

Region	1990	2000	2005	2010	2015	Annual change 1990-2015	Annual change 2005-2015
	1 000 ha						
North Europe	69,975	70,852	70,736	70,781	70,832	34	10
Central-West Europe	35,021	36,385	37,162	37,861	38,582	142	142
Central-East Europe	41,628	42,762	43,300	43,873	44,494	115	119
South-West Europe	24,852	28,705	29,353	30,531	30,913	242	156
South-East Europe	26,297	27,447	28,429	29,562	30,446	166	202
Europe	197,773	206,151	208,980	212,607	215,267	700	629
EU-28	147,956	154,740	156,758	159,236	160,931	519	417

Small scale planting of trees on agricultural land

Trees as landscape features, including field trees, hedging and tree lines or groups along field edges, exist across agricultural land. Under existing policy landscape feature retention and addition can be supported by a variety of different CAP measures. This is intended to promote the delivery of wider ecosystem services such as prevention of erosion, increasing soil organic matter and the provision of habitats. While such features may be managed and material will arise from these, they are not being actively grown for their woody biomass. Harvesting of such biomass may occur during maintenance activities but this would normally be considered an agricultural or landscape management residue.

2.3.4 Biomass from primary wood production

In 2015 forest area in the EU28 amounted to approximately 161 million ha and accounted for 38% of the total land area (

Figure 2-13). Other wooded lands covered an additional area of 21 million ha (5%). The distribution by forest type varies considerably by region; 45% of European forests are predominantly coniferous, 36% are predominantly broadleaved and the remainder are mixed (Figure 2-14) (Forest Europe, 2015)[31].

The area of semi-natural forest and plantations increased in continental Europe over the 20-year period 1995-2015. Around 87% (174 million ha of the forest area) of European forests were classified as semi-natural in 2015. Undisturbed forests cover around 3% (7.3 million ha) and plantations 9% (12.9 million ha) of forest area in Europe (8% for the EU 28[32]). The highest share of undisturbed forests within the forest area can be found in countries of Central-East

[31] It should be noted that where possible data is set out for EU28, but wider 'European' statistics include data reported from European countries outside the EU for example Iceland.
[32] https://www.eea.europa.eu/soer-2015/europe/forests

and South-East Europe, while the share of plantations is the highest in the Central-West, South-West and South-East European regions (Forest Europe, 2015).

Over the last 15 years, the area of forest in Europe designated for biodiversity and landscape protection increased by half a million hectares annually (Forest Europe, 2015). Around 12.2% (or 29.9 million ha) of European forests are protected with the main objective of conserving biodiversity (Figure 2-15). Around 7% have the protection of landscapes representing an area of 19 million ha as a main objective. The strictness of protection for biodiversity varies considerably within Europe: while restrictive protection with minimal or no intervention dominates in North Europe and some East European countries, active management in protected areas is more common in Central and South European countries.

Commercial wood volume produced by felling in the EU mostly comes from Sweden, France, Germany, Finland, Poland and Austria, which amounted to approximately 330 million m^3 wood over bark in 2010, i.e. 68% of the total EU-27 (Eurostat, 2011). According to the same source, over two thirds (i.e. 67.9 %) of wood removal from felling was from coniferous species and almost a third from broad-leaved species (i.e. 32.1%). The coniferous species most often exploited in the EU are Scots pine, Norway spruce and Sitka spruce, and to some extent Lodgepole and Maritime pine, Larch and Douglas fir. Among the broadleaved trees the species most often used are Birch, Aspen, Alder, Eucalyptus and Oak (ICNF, 2013; Swedish Forest Agency, 2015).

Figure 2-13 Extent of forest and other wooded land, 2015[33]

Source: (Forest Europe, 2015)

Region	Forest....		....of which available for wood supply			other wooded land	
	1000 ha	% of total land	1000 ha	Total forest area of countries reporting FAWS 1000 ha	% of forest land	1000 ha	% of total land
North Europe	70,832	53.2	55,223	70,832	78.0	5,874	4.4
Central-West Europe	38,582	27.6	36,290	38,582	94.1	918	0.7
Central-East Europe	44,494	27.1	31,019	44,085	70.4	914	0.6
South-West Europe	30,913	35.0	25,016	30,897	81.0	12,747	14.4
South-East Europe	30,446	23.5	18,391	24,826	74.1	15,308	11.8
Europe	215,267	32.8	165,939	209,223	79.3	35,760	5.5
EU-28	160,931	37.9	134,486	160,931	83.6	20,987	4.9

[33] To note there is no harmonised definition of forest available for wood supply in Europe see article https://www.unece.org/info/media/presscurrent-press-h/forestry-and-timber/2016/unece-and-fao-call-for-a-harmonized-assessment-of-forest-availability-for-wood-supply-in-europe/doc.html It is meant to consider and exclude areas protected for environmental purposes and social restrictions. Not wider issues such as economic restrictions on capacity

Figure 2-14 Proportion of forest area by forest type

Source: (Forest Europe, 2015)

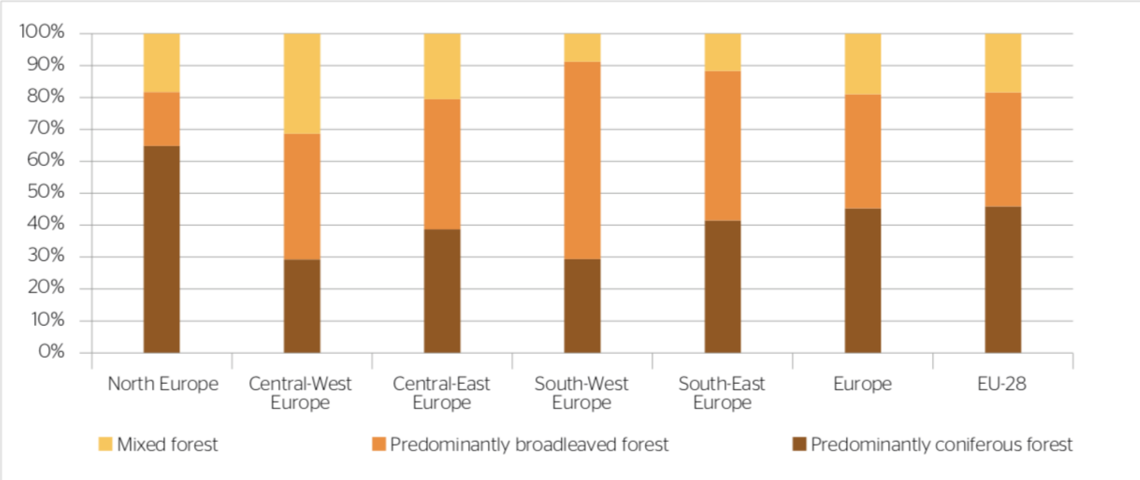


Figure 2-15 Area of forest and other wooded land protected and percentage of protection for biodiversity and landscape in Europe, 2015

Source: (Forest Europe, 2015)

	Europe	
Management objective	million ha	% of total forest area
Biodiversity, MCPFE categories 1.1-1.3	29,9	12.2
1.1 No active intervention	3,6	1.5
1.2 Minimum intervention	7,3	3.1
1.3 Conservation through active management	19,0	7.6
Landscape, MCPFE Category 2	17,3	7.2
Total - Biodiversity and Landscape combined	47,2	19.4

Use of woody biomass for energy

EU forestry statistics tend to focus on the economic factors surrounding forests, total area and forest increment, rather than providing detail of the exact material extracted and what this may consist of. Therefore, statistics here primarily reflect data from industry (in the form of (AEBIOM, 2017) statistics) and key synthesis of data on biomass for energy including the Task 1 report for RECEBIO (led by industry experts INDUFOR) who gathered data on roundwood and wood use in energy (Pekkanen et al, 2014). It synthesises this with more recent analysis on forest biomass by the JRC (Camia et al, 2018).

Some efforts have been made to model the volume of used forest compared to unused forest. Under the Recebio I study (Forsell et al, 2016a), used and unused forest figures based on the G4M model (which assesses the extraction of forest material based on economic parameters) was used as a mechanism for interpreting intensity of forest use. In 2010 the area of used forest was estimated to be 105 million ha (69% of area) and unused forest was 48 million ha in 2010 (31% of area)[34].

Analysis of use of woody biomass focuses on the consumption of biomass volumes in the form of wood flows, comparing this to total stock of material and annual increments in growing stock (measures in Mm^3). Wood flow analysis examines the level of extraction of wood and the uses to which it then 'flows' allowing proportions being used for energy to be estimated.

Analysis by Indufor (2013) estimated that the total wood raw material use was 942 million m^3 RWE (roundwood equivalent). About 31% (approximately 292 million m^3) of that was used by the bioenergy sector. However, only 143 million m^3 was considered primary production, whilst the remaining 150 million m^3 was from industrial side streams. The EUWood study (Mantau et al, 2010) estimated the total wood consumption at 825 million m^3, of which energy use accounts for 45% of wood harvested (Figure 2-16).

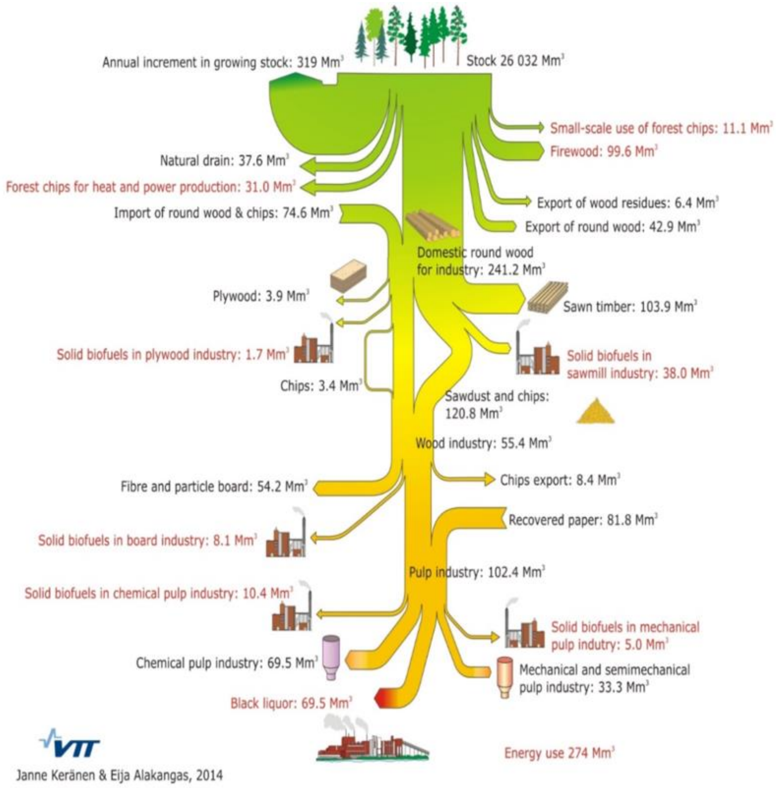
Keränen & Alakangas (2014) reproduced in AEBIOM (2017) also analysed wood flows in Europe (**Error! Reference source not found.**. Their analysis estimated that domestic r oundwood for industry use and exported roundwood was approximately 284 million m^3. The total wood used for energy is 274 million m^3. Energy wood from primary sources, i.e. firewood and forest chips covered about 142 million m^3 (approx. 51%) of total. The remaining, approximately 132 million m^3 (approx. 48%) of wood used for energy was a result of by-product flows of wood consuming industries.

Analysis by JRC (Cazzaniga et al, 2019) compiled data on past wood extraction and use, reviewing: total wood use, the split between energy and materials, and the use of both direct wood and indirect wood sources for energy (

Figure 2-17 Wood flows in Europe (2014)

Source: Keränen & Alakangas (2014) reproduced in AEBIOM (2017)

[34] Used forest is defined in the analysis as - Forests that are used in a certain period to meet the wood demand are modelled to be managed for woody biomass production. This implies a certain rotation time, thinning events and final harvest. Examples of used forests are:
- A forest that is actively managed (through thinning or clearcut activities etc.) on a regular basis and the wood is collected for subsistence use or to be sold on markets.
- A forest used on a regular basis for collection of firewood for subsistence use or to be sold on markets.
- A forest concession or community forest used for collection of wood for export and/or domestic markets.
Unused forest is defined in the analysis as – Forests that currently do not contribute to wood supply (for economic reasons) as determined by the model. However, these forests may still be a source for collection and production of non-wood goods (e.g. food, wild game, ornamental plants).



Source: Janne Keränen and Eija Alakangas, VTT 2014

74 of 264

Figure 2-18). Within the JRC data (noting the limitations identified in the study) the following trends are suggested:

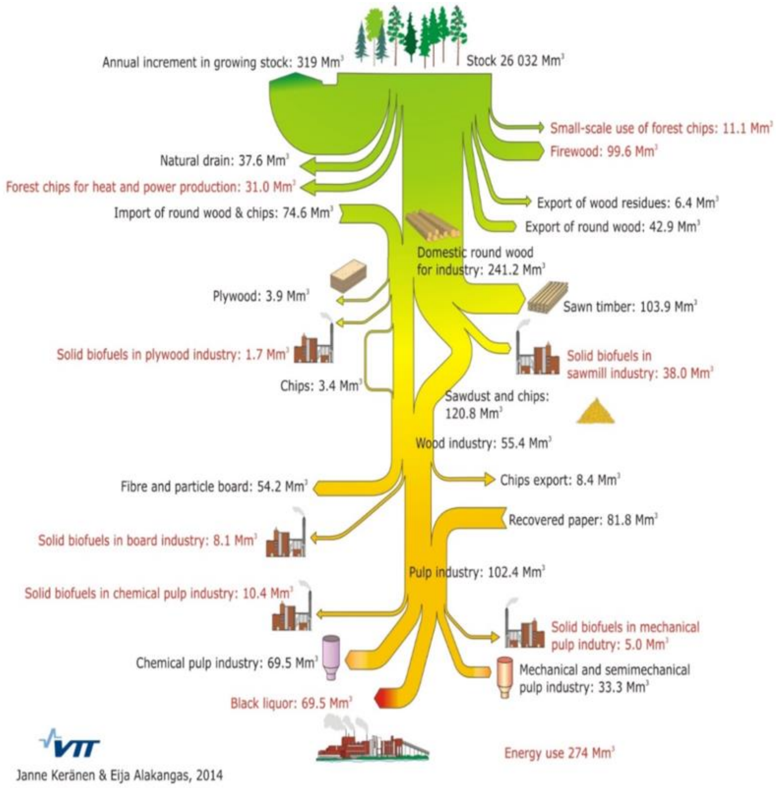
- that total wood use in the EU 28 is increasing (rising from 737Mm^3 in 2009 to 921Mm^3 in 2015), this includes a limited volume of imports, but the majority is from EU forests;
- that the percentage of total wood use for energy has increased, as has the total volume of wood used for energy (rising from 338Mm^3 in 2009 and 45.8% of use to 451Mm^3 in 2015 and 49%). However, as noted in the JRC analysis this also reflects transformational changes in the material using sectors linked to economic downturn in 2008 and wider socio-economic conditions i.e. decline in construction, decline in pulp and paper industries;
- that the proportion of direct wood i.e. biomass entering energy production without further treatment is declining while the proportion of indirect wood used for heat and power are increasing. Direct wood encompasses the majority of 'primary biomass extraction for energy' including wood chips. However, some of the indirect wood bracket would likely also include primary biomass from forests i.e. thinnings and other forest residues used for pellet production. Therefore, these categories cannot be used directly to determine the split between use of primary forest/woodland biomass including forest residues and thinnings and secondary materials from processing residues and post-consumer waste.

These studies give variable estimates of the proportion to wood harvested from EU forests being used for energy production. The studies cited above estimate between 31% and 49% of wood is used being for energy[35] (see Figure 2-16 and

Figure 2-17 Wood flows in Europe (2014)

Source: Keränen & Alakangas (2014) reproduced in AEBIOM (2017)

[35] This is consistent with more recent findings, for example in the 2015 Joint Wood Energy Enquiry of 38.2% of the wood fibres for energy generation derive directly from woody biomass from forests and wooded areas outside forests http://www.unece.org/fileadmin/DAM/timber/wood_energy/JWEE2015-brief-analysis.pdf. Data reveals the trend that member states increasingly source wood energy from indirect sources (e.g., wood residues and processed wood-based fuels) and less wood directly from the forests. Also, wood energy use in the industry and residential sector decreased while the power and heat sector consumed more wood for energy.



Source: Janne Keränen and Eija Alakangas, VTT 2014

74 of 264

Figure 2-18). However, around 50% of this energy (48% in Keränen & Alakangas and 51% in Indufor’s analysis) is produced using primary wood i.e. material that has not already been utilised or become a waste stream from an industrial process. Analysis by JRC (Camia et al, 2018) has identified, however, that the split between wood for energy and material use varies considerably between EU Member States. This is supported by analysis of the volumes of roundwood extracted directly and used for energy, which also vary considerably by Member State (

Figure 2-19 and

Figure 2-20).

Figure 2-16 Wood demand by wood consuming industries by EUWOOD and INDUFOR

Source: Indufor (2013)

Study	**EUWood**	**Indufor**
Reference year	2010	2011
	Million m³	
Total Wood Consumption	825	942
Total Material Use	*457*	*649*
Wood Products Industry	314	308
Pulp and Paper		341
Pulp	143	
Total Energy Use	*368*	*293*
Wood products industry side streams		150
Primary energy use		143
Energy use, %	*45%*	*31%*
Material use, %	*55%*	*69%*

Figure 2-17 Wood flows in Europe (2014)

Source: Keränen & Alakangas (2014) reproduced in AEBIOM (2017)

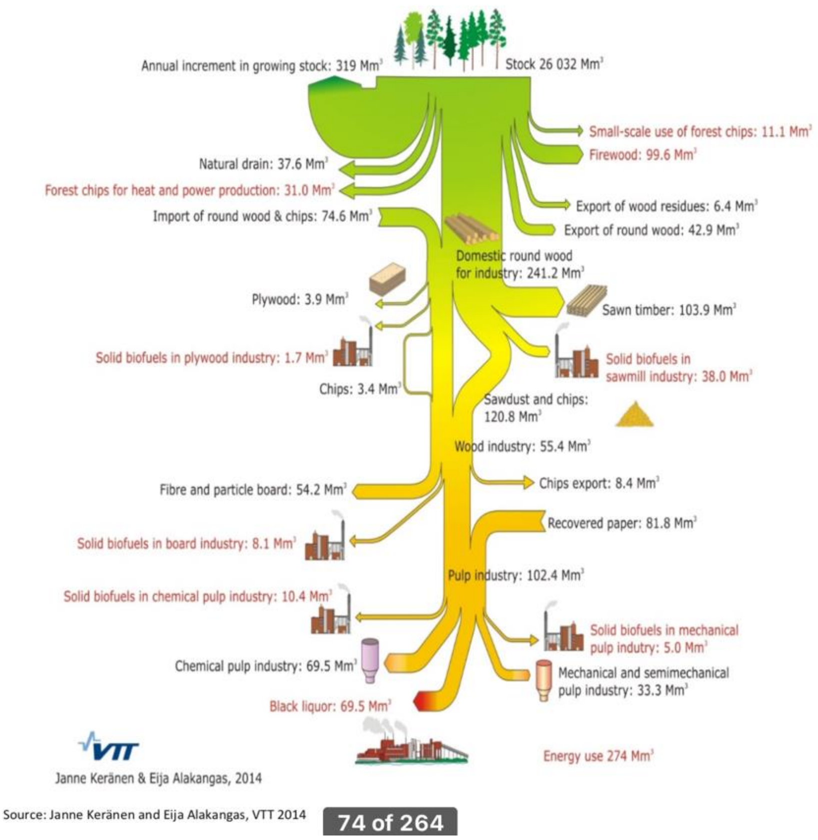


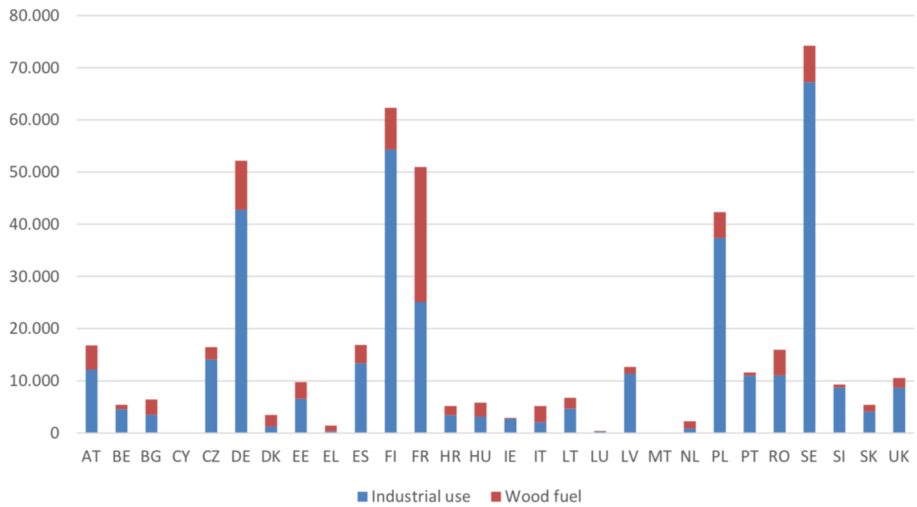
Figure 2-18 Data on wood use for energy – 2009 to 2015

Compiled from JRC (Cazzaniga et al, 2019) – Summarising total wood use by the EU 28 proportions of wood used for energy vs materials and the proportions of wood use that is direct wood for energy compared to indirect sources. Note that primary energy use figures will include all wood used as primary material i.e including primary forest residues.

Year	Total wood use EU 28 (1000m3)	Wood Use - Energy		Wood Use - Materials		Direct Wood[36] – Heat and Power		Indirect Wood[37] – Heat and Power	
		Volume (1000m3)	Percentage of total wood use	Volume (1000m3)	Percentage of total wood use	Volume (1000m3)	Percentage of total wood use	Volume (1000m3)	Percentage of total wood use
2009	737,218	337,568	45.8	399,650	54.2	142,769	19.4	153,552	20.8
2010	789,652	358,035	45.3	431,617	54.7	152,018	19.3	157,763	20.0
2011	810,131	369,408	45.6	440,723	54.4	158,647	19.6	165,741	20.5
2012	836,368	394,549	47.2	441,819	52.8	164,555	19.7	182,398	21.8
2013	876,488	426,128	48.6	450,360	51.4	165,047	18.8	200,116	22.8
2014	898,346	435,936	48.5	462,411	51.5	163,598	18.2	208,809	23.2
2015	920,826	451,082	49.0	469,744	51.0	165,930	18.0	222,287	24.1

Figure 2-19 Roundwood removals in EU28 Member States according to end use in 2016 (1000m3)

Source: (AEBIOM, 2017)
To note that these figures are only for roundwood which is generally distinguished from pulp wood i.e. lower grade wood, see data below.



[36] Any wood fibre entering energy production without any further treatment or conversion. It comprises removals from forests and outside.
[37] Processed and unprocessed co-products (residues) from the wood processing, solid (sawdust, chips, slabs, etc.) or liquid from the pulp industry (black liquor or tall oil). Processed wood fuels with improved energy content per bulk volume (compressed), such as wood pellets, briquettes but also wood charcoal are also included under indirect supply. Moreover, it includes the post-consumer recovered wood: any waste wood fibre after at least one life cycle. It comprises wood from construction, renovation and demolition, but also packaging as well as old furniture.

Figure 2-20 Wood removals from forests in EU28 Member States by assortment in 2016 (thousand m3)

Source: (FAOSTAT cited in AEBIOM, 2017)

	Roundwood	Wood fuel	Sawlogs and veneer logs	Pulpwood	Other industrial roundwood
EU28	452.375	97.677	196.005	149.179	9.515
AT	16.763	4.590	9.006	3.167	0
BE	5.412	893	2.965	1.381	173
BG	6.410	2.928	1.502	1.919	61
CY	16	13	3	0	0
CZ	16.472	2.390	9.074	4.918	90
DE	52.194	9.413	28.183	11.992	2.605
DK	3.456	2.218	842	333	62
EE	9.735	3.161	4.076	2.445	52
EL	1.432	1.065	304	0	63
ES	16.880	3.500	3.943	9.167	270
FI	62.291	7.964	23.410	30.917	0
FR	50.971	25.859	16.468	8.128	516
HR	5.165	1.768	2.402	988	6
HU	5.798	2.679	1.105	1.144	871
IE	2.908	203	1.559	982	164
IT	5.153	3.064	1.120	682	287
LT	6.747	2.085	3.494	1.168	0
LU	381	70	113	198	0
LV	12.651	1.300	7.969	2.640	742
MT	0	0	0	0	0
NL	2.271	1.397	392	470	13
PL	42.335	4.975	16.858	19.455	1.047
PT	11.613	600	2.054	8.635	324
RO	15.927	4.882	8.993	1.045	1.008
SE	74.200	7.000	35.700	31.000	500
SI	9.267	515	5.003	3.717	32
SK	5.381	1.272	2.989	990	131
UK	10.545	1.872	6.477	1.698	498

Source: FAOSTAT

Forestry Residues – Estimates

In the context of this analysis, forestry residues are considered to be the use of primary biomass i.e. stumps, thinning, tops and branches, rejected saw logs and other low value trees that are considered to be residual to the primary production of roundwood. This review does not consider the residues emerging as by-products from wider industrial wood processing, which are noted to account for approximately half of biomass wood-based energy (see above).

Few studies look in detail at the quantity of primary residues being removed from European forests. Tending to look at either the broader increment removals or the products associated with the use of residues. In their 2018 analysis of biomass production, supply, uses and flows in the European Union, JRC estimate that over the period 2004-2013 the yearly average felled was 281Mt of which 224Mt were removed and 27Mt (20%) were left in the forest as logging residues (Camia et al, 2018). However, they also note that reported removals have been shown to be underestimated up to 20%.

Given the challenges of quantifying such data, studies tend to look at the products associated with the use of residues. In the case of primary residues from forest management the main product produced is wood chip for energy (although studies note that some roundwood may also be used). Although increasingly low value wood i.e. rejected sawlogs, thinning and undersized trees are also being utilised for pellet production (where such facilities exist) –

studies have suggested that for wood pellets the source is primarily industry co-products in Europe (approximately 74%) (UNECE/FAO, 2015). Hence use of wood chip and, to a lesser extent wood pellets, can currently be used as a proxy to identify volumes/trends in use of primary residues.

Forest chips are fresh wood chips made of wood being harvested directly from the forest, used for energy production, and has not had any previous industrial use. Use of forest chips made of tops and branches and small diameter stemwood is tightly integrated in industrial roundwood supply. Forest chip raw material is also collected from thinning operations in young and mid aged forest stands aside with pulpwood harvest (Forsell et al, 2016a).

Use of forest chip has been increasing in the EU27, estimated as approximately 45 million m3/a in 2011 (Pekkanen et al, 2014). Major consumers of forest chips in the EU in 2011 were Sweden (10.5 million m3/a), Germany (9.0 million m3/a), Finland (7.5 million m3/a) and Italy (6.3 million m3/a). The data illustrate a strong increasing trend in the consumption of forest chips for the whole EU between 2007 and 2011 (Figure 2-23).

It is not possible to determine the exact proportion of forest residue use nor attribute proportions to bioenergy use. However, it is possible to presume that the use of primary forest residues is increasing - based on the consumption of wood chip that utilises this as a primary product. It should be noted, however, that a proportion of this shift will be from traditional use of forest primary residues as logs and other sources of traditional energy. Despite an extensive review of literature, defined levels of extraction for different types of residues have not been identified. The EU wood study (Mantau et al, 2010) does make assumptions about the level of residues linked to forest extraction, noting that about 52% of the total potential is in stems, while logging residues and stumps represent 26% and 21%, respectively (Figure 2-21). Other biomass, i.e. stem and crown biomass from early thinnings, represent only 1% of the total potential.

EU wood analysis (Mantau et al, 2010) estimated the theoretical forest biomass potentials based on the EFISCEN model based on different mobilisation scenarios. For each mobilisation scenario the extent of different forestry sources was estimated (Figure 2-22). However, comparable data looking at actual practices and extraction rates in forest has not been identified systematically.

Figure 2-21 Estimated potentials from EU (27) forests per year and the types of material anticipated to be available

Source: Mantau et al (2010)

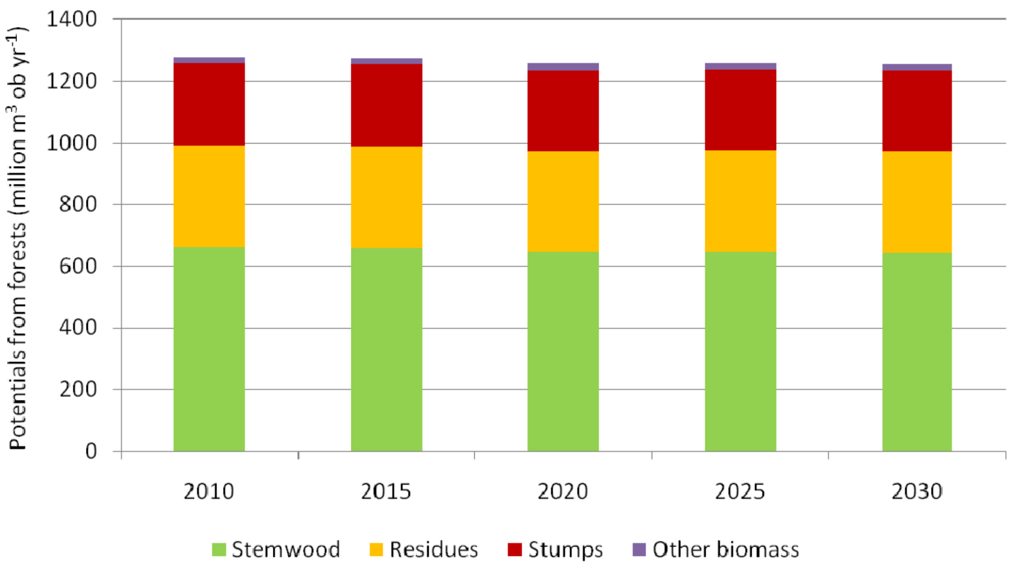


Figure 2-22 Comparison of biomass potentials from forests in EU-27 for different mobilisation scenarios in 2010 and 2030

Source: (Mantau et al, 2010)

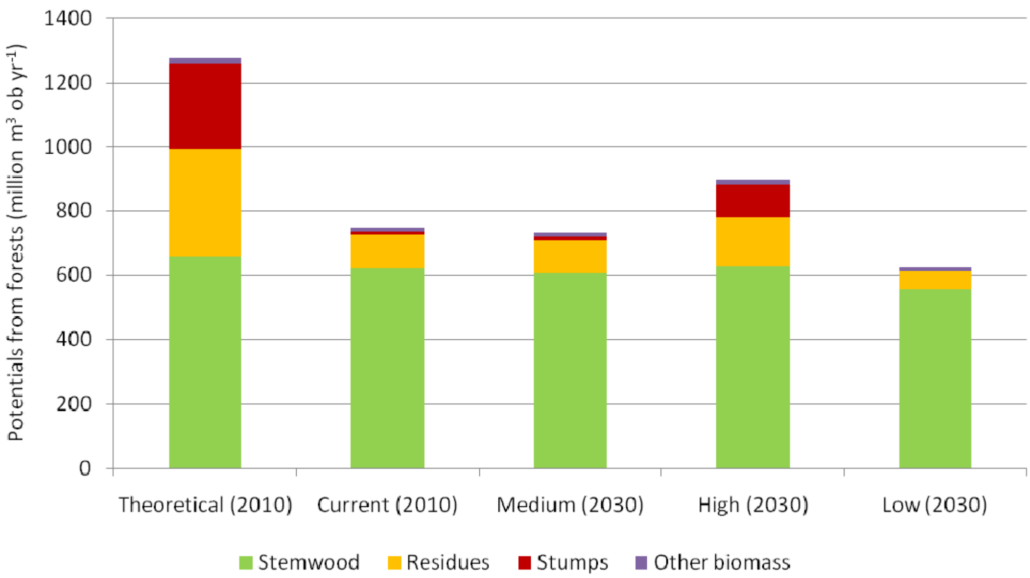
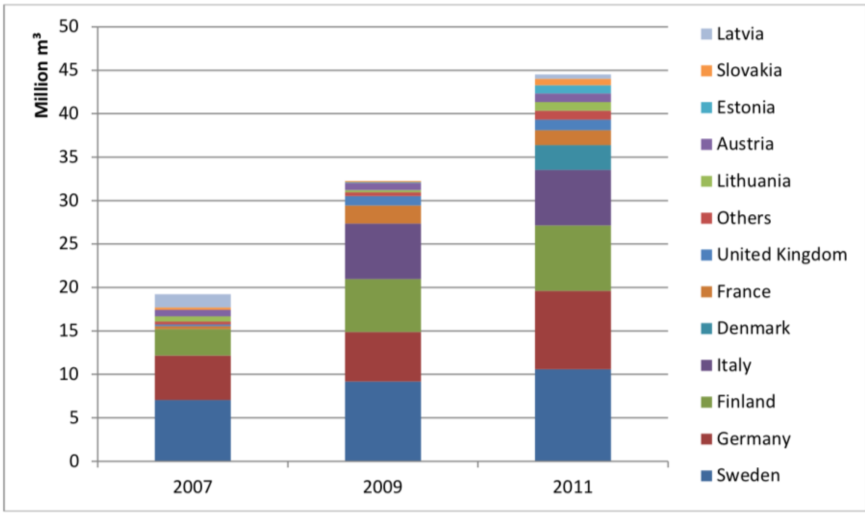


Figure 2-23 Consumption of forest chips in the EU 27

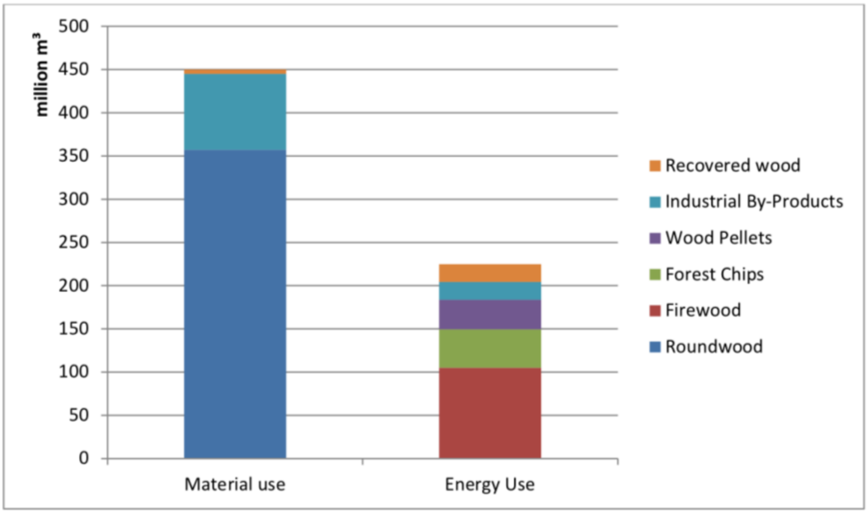
Source: Compiled by INDUFOR for RECEBIO, Task 1 (Pekkanen et al, 2014)



Source: National Statistics (Sweden, Finland), Joint Wood Energy Enquiry. Holzmarktbericht. Data missing for Romania, Slovenia, Spain.

Figure 2-24 Estimated current consumption of wood biomass for material and energy use in the EU27 (2011)

Source: Compiled by INDUFOR for RECEBIO, Task 1 (Pekkanen et al, 2014)



Source: Material use: Indufor, Study on the Wood Raw Material Supply and Demand for the EU Wood-processing Industries. Excludes recovered paper and imported pulp (RWE). Energy use: Data used in section 2.2.3.

Future use of woody biomass

60% of European countries reporting to Forest Europe (Forest Europe, 2015) have explicit targets for increasing domestic wood production intended to meet the increasing demand in both material and energy use (i.e. Finland, Germany) and reducing the trade deficit (France). One third of these countries also report quantitative objectives for greater energy use in the range of 20% to 60%. The countries referred to increased mobilisation (Finland, France), increased productivity (Sweden), increased forest area (Ireland) and increasing managed area (Hungary). This reflects both the prioritisation of biomass for energy, but also wider promotion of biomass products within the economy (i.e. efforts to promote the Bioeconomy).

When considering the future demand for woody biomass for energy, two studies using Globiom modelling, RECEBIO I and RECEBIO II, consider the impacts of using solid, wood biomass in particular in the heat and electricity sectors (see Annex 2 for further details of these studies' findings). These studies estimated the shift in the area of used and unused forest, the expansion in forest area and proportion of wood extracted used for bioenergy.

Under Recebio I (Forsell et al, 2016a) analysis showed an increase in the area of used forest under the business as usual (BAU) scenario from 105Mha in 2010 to 116Mha in 2030 and 124Mha in 2050. This is primarily driven by afforestation of areas of 'other natural land'. There is also noted to be a decline in the area of unused forest from 48 in 2010 to 42Mha by 2050. Recebio II (Forsell et al, 2016b) used as its basis a revised set of figures for energy demand from biomass. Under Recebio II the BAU scenario identified 130Mha of used forest exist in 2050, again this expansion (35Mha split between SRC expansion and afforestation of land conversion between 2010 and 2050) comes at the expense of other natural land reducing this land category by more than 50%. It should be noted under scenarios that exhibited higher demand for biomass from energy, conversion of unused to used forest land was seen. In both studies an expansion in domestic wood production was seen, at a rate greater than could be accounted for by afforestation increase. The studies conclude that this implies increasing intensity of wood supply and forest management.

Wood flow analysis under Recebio I identify further patterns in terms of wood consumption. Wood consumption for both energy and material uses increases over the period (from 306Mm3 in 2010 to 419Mm3 in 2050 for energy and 302Mm3 in 2010 to 379Mm3 in 2050 for material uses). This implies expanded wood demand overall and a shift in the proportions being used for energy – in 2010 approximately 50% is estimated to be used for energy; in 2030 approximately 54% is used for energy and in 2050 53% is used for energy. The balance between use of industrial products compared to primary material remains between 50 and 55% over the period[38].

[38] Data based on wood flows see P37 of Recebio I task 3 report.

2.4 Summary and synthesis of evidence regarding bioenergy feedstock production in the EU

While there are considerable data limitations, it is necessary to develop a baseline against which habitat and species sensitivity and responses can be assessed to analyse bioenergy's impact in terms of exposing protected habitats and species to risk. The following section represents an estimate, based on the available evidence presented above, of biomass feedstocks being produced in the EU for energy generation. It should be noted that these figures are purely for the purpose of the vulnerability assessment of habitats and species within the context of this analytical exercise and should not be reproduced for other purposes. They should not be added together or considered to provide a picture of European land use for energy. Moreover, there are inherent uncertainties within all figures associated with bioenergy feedstocks being part of a continuum of production of forest and agricultural biomass. This continuum means that uses and feedstocks are often interchangeable and substitutable dependent on prices and demand signals, global production outputs and flux in oil prices. The following tables therefore supply best estimate snapshots of the 2015 baseline and projections for 2030 based on currently available data.

It should be noted that 2015 is considered to represent the current status of pressures in Europe and a point at which data are sufficient to make the relevant assessments, However, as outlined above, binding targets promoting use of renewable energy and driving biomass for energy demand, have been in place since 2009. Therefore, to understand the nature of EU biomass demand, excluding the current EU targets a base year of 2008 (in line with baselines for the RED sustainability criteria) would need to be examined. However, it should be noted that before this period bioenergy was already being promoted by Member States linked to rural development and national renewable energy goals. Moreover, traditional uses of woody biomass for heating have long been integrated with many European forestry and rural land management systems.

2.4.1 Summary of trends and key messages of relevance to protection of habitats and species under the Habitats and Birds Directives

The current overarching trend is for the use of biomass for energy to increase, it is expected to continue to do so up to 2030 driven by demand associated with policies to promote renewable energy. However, the usage patterns of different feedstocks are anticipated to evolve given strengthening and extension of rules on the sustainable sourcing of biomass feedstocks used for energy (both a result of changes adopted in the 2015 amendments to the RED to adapt to ILUC concerns and new rules on agricultural and forestry feedstocks applicable post 2020). These pressures will interact with existing alternative uses and demand for materials to define the bioenergy feedstock consumption patterns in the next decade.

Conventional crops used for biofuels for transport, bioliquids and biogas generation have been seen to expand because of the EU's Renewable Energy Directive and are currently estimated to account for around 5% of arable land. While still predicted to grow to a limited extent to 2020[39], up to 2030 the total area used for cultivation of feedstocks produced from conventional crops is anticipated to remain relatively static (see Figure 2-26). A slight decline

[39] Agricultural Outlook 2018-2030, European Commission

in conventional crop-based feedstock use for bioethanol and biodiesel is matched by a slight increase in use of crops for biogas. In the case of biogas, food and feed crop sources will be covered by rules on sustainable sourcing of biomass from 2020, and Member States are noted to be evolving support policies for biogas (to address concerns over in particular use of maize). However, existing plants often hold long term contracts limiting likely feedstock transition during their lifetime (up to 20 years) and additional plant are still anticipated, with an increasing emphasis on renewable gas as a resource up to 2050.

It should be noted that while the overarching data sets for the EU suggest a relatively static level of usage of conventional crop feedstocks for bioenergy up to 2030, the locations in which crops are grown are likely to evolve. For example, existing data show levels of green maize cultivation in Estonia and Latvia significantly expanding, while it is declining or static elsewhere. The same is true for sugar beet, for which production has declined significantly in Portugal but increased for example in Romania. Localised pressures will remain, potentially putting pressure on semi natural habitats. Moreover, there is evidence of intensification of production of potential bioenergy feedstocks including sunflower cultivation, other oil crops and sugar beet. This may affect species dependent on more extensive agricultural systems, although only a portion of such intensification can be attributed to bioenergy demand.

At present dedicated energy crops are produced on relatively limited areas in the EU (equivalent to less than 0.5% of arable land even when higher estimates of current cultivation are taken into account); looking to the future these crops are expected to expand. Such crops can be used for multiple energy end uses and can deliver over a relatively short timeline (compared to forestry investments). Estimates of anticipated future production vary widely; this analysis has adopted relatively conservative estimates of expansion identified in scenarios for future demand to 2030. This choice was made given that, to date, there has been considerable inertia in terms of the adoption of dedicated energy crops including Short Rotation Coppice (SRC) and Miscanthus (both used as proxies for energy crops within the analysis). However, despite this relatively conservative approach, we identify an estimated expansion of SRC from 10,000 ha in 2015 to 2,500,000 ha in 2030. We also estimate an equivalent expansion in Miscanthus for bioenergy (see Figure 2-26).

Energy crops require land for production, and there have been extensive discussions as to their siting. SRC is often expected to expand into non arable land, with models predicting expansion into semi natural areas or grasslands. Meanwhile more formal ‘energy crops’ such as Miscanthus are predicted to be placed on arable land. However, there has been discussion about the ILUC consequences of expansion onto croplands. This has led to suggestions that energy crops should be located exclusively on ‘spare’ or ‘marginal’ land including less productive land and abandoned agricultural land. ILUC is a discussion around the displacement of impacts intra EU and globally associated with land use change, including biodiversity consequences. However, such spare or marginal land in Europe is often semi natural in character and managed at lower intensities. There is a risk that energy crops become concentrated in these semi natural areas. For example, the relatively conservative estimates of SRC and Miscanthus area considered in this analysis to 2030 would be equivalent to approximately 10% of the EU's current area of permanent pasture[40].

Estimated wood flows by JRC (Camia et al, 2018) for 2010 to 2015 show a significant increase in wood use over the period. The total use of wood for both bioenergy and materials is seen to be increasing; however, the proportion of wood being used for energy also increased, meaning a greater proportion of total wood is being used for energy. In 2015 it is estimated that 24.8% of primary wood material extracted from EU forests is used directly for bioenergy (the total bioenergy demand for wood is higher than this, but the remainder is provided for by secondary wood sources including residues from material uses). Afforestation over the same period was 1,695 (1000 ha) or approximately a 1% increase in total forest area. Of this only a proportion will be linked to increased pressure associated with bioenergy, as part of wider decision making around forest establishment. However, the differences between the proportion of new forest (1%) and the proportion of additional woody biomass extracted between 2010 and 2015 (16% increase, of which 71% was attributed to bioenergy end uses) suggest an increasing intensity of forest management to deliver the increased volumes of wood.

Looking to the future use of wood, and associated impacts on forestry, there are significant uncertainties in the level of total wood use anticipated. This is in part due to the increase in wood use between 2009 and 2015 being more significant than scenario models had predicted. However, there are some common trends indicated across studies. Firstly, that total wood use for both bioenergy and material use will continue to increase. Bioenergy use of woody biomass will increase in terms of both the volume of wood use and the proportion of total wood demand dedicated to bioenergy; this is expected to exceed 50% by 2030, resulting in bioenergy use being a higher usage category than material use. The proportion of bioenergy from secondary sources will decline if bioenergy becomes the dominant wood use, as the proportional availability of such resources will be less given they emanate from material flows. Therefore, in 2030, bioenergy use of primary forest material is anticipated to rise to 29.7% of total wood use.

When considering biomass feedstock use it is important to note their interdependency. Sensitivity analysis completed on the level of demand (within the RECEBIO project), for example from SRC, notes that non delivery of that portion of biomass would result in additional pressures on other sources to deliver renewable energy ambitions. This includes potentially increased forest extraction rates in the EU or in third countries.

2.4.2 Estimated exposure to risk factors associated with bioenergy feedstocks currently

Based on the information in section 4.1.2, the best estimate is that conventional bioenergy feedstocks are currently cultivated on 2.8% of a total of 218 million ha of arable, grassland and shrubland. The total area of energy crops including Miscanthus and SRC is estimated at 0.01% of arable, grassland and shrubland, and this is split between the arabla and non-arable areas. Based on these estimates, and described in Figure 2-25, we have derived the following

[40] It should also be noted that afforestation also commonly is anticipated to take place on such land again adding to a potential cumulative pressure.

estimates of areas currently exposed to risk factors associated with bioenergy crops on arable, grassland and shrubland:

- Conversion of permanent grassland to conventional crops for bioenergy on 0.07% of permanent grassland.
- Intensification of management on 0.005% of grassland.
- Intensification of cropland through changes in crop type on existing arable land, principally arable crops to maize on 1% of arable land.
- Removal of residues from crops (straw etc) on 1.5% of arable land.
- Cultivation of energy crops such as miscanthus and SRC on an area equivilant to 0.01% of arable, grassland and shrubland.
- Afforestation of grassland or shrubland on an areas equivalent to 0.047% of arable, grassland and shrubland.
- Afforestation of arable cropland on an area equivalent to 0.0025% of arable, grassland and shrubland.
- Small scale planting of trees in farmland on an areas equivalent to 0.005% of arable, grassland and shrubland.

The best estimate is that 24.8% of the total wood use in Europe in 2015 was used for bioenergy, i.e. 0.8% of the total standing stock of forests.

See Figure 2-25 and section 4.1.2 for details of the assumptions and method used.

2.4.3 Estimated exposure to risk factors associated with bioenergy feedstocks in 2030

The proportion of land under conventional bioenergy feedstocks is not expected to change, so the assumption remains that conventional bioenergy feedstocks are cultivated on 2.8% of arable, grassland and shrubland in 2030. The estimates of areas exposed to risk factors associated with bioenergy crops on arable, grassland and shrubland in 2030 are:

- Removal of residues from crops (straw etc) on 14% of arable land.
- The combined area for energy crops i.e. Miscanthus and SRC in 2030 is estimated to be 5.3 Mha dedicated to meeting bioenergy demand. This is equivalent to 2.4% of arable, grassland and shrubland.
- Conversion of grassland or semi natural habitats to Miscanthus etc or SRC on the equivilant to 1.1% of arable, grassland or shrubland.
- Conversion of arable cropland to Miscathus etc or SRC on the equivalent to 1.3% of arable, grassland and shrubland.
- Afforestation of grassland or other semi natural habitats on the equivalent to 0.21% of arable, grassland and shrubland.
- Afforestation of arable cropland on the equivalent to 0.011% of arable, grassland and shrubland.

The best estimate is that 29.7% of the total wood use in Europe will be used for bioenergy in 2030, i.e. 1.2% of the total standing stock of forests in 2015.

Figure 2-25 Estimates of the current use of biomass feedstocks for energy

To note – extended text included to explain workings for this draft to facilitate discussion with the European Commission regarding the numbers reached as a basis for further assessment. Table text will be streamlined following discussion and agreement with the European Commission on the final assessment outcomes.

NB. Text and figures in bold indicate assumptions and estimates that are used in the semi-quantitative vulnerability assessment in chapter 4.

	Evidence	Confidence	Calculation Method	Estimates
Conventional crops (i.e. food and feed/forage crops)	Current areas of bioethanol, biodiesel and biogas feedstocks being cultivated and potential areas i.e. areas of wheat, oil seeds, sugar beet, green maize compared to areas estimated in use for bioenergy. Based on It should be noted that while grass can be used as an energy feedstock this is not analysed in detail here. This is because grass use to date is normally integrated within feed cycles or wider management cycles. It is, therefore, not possible to definitively distinguish activities and use patterns. Figure 2-1 unless otherwise stated. Figures in 1000 ha, utilising a 2015 base year used for consistency Area rapeseed in Europe – 6,500 thousand ha. 43.4% estimated to be used for bioenergy.	Medium – area data for some elements estimated. Data on areas and proportions are from the same base year, however, the data are from different reports. Hence there may be a question of consistent baselines. Although data on proportions of crops are based on Agri Outlook estimates, utilising Eurostat data baselines. Data on Sunflower areas less reliable in terms of proportion used for energy vs other sources.	Based on 2015 base year Area Rapeseed for bioenergy estimated as – 3140 (1000ha) Area Soybean for bioenergy estimated as – 45 (1000ha) Area of Sunflower for bioenergy estimated as – 420 (1000ha) Area of wheat for bioenergy estimated as – 1115 (1000ha) Area of sugar beet for bioenergy estimated as – 179 (1000ha)	Land utilised to cultivate conventional feedstocks in Europe – 6.05 million ha; approximately 5.7% of arable land (based on 107 million ha), 3.6% of arable land and grassland in 2015[41], (166 million ha) and **2.8% of the 218 million ha of arable, grassland and shrubland** (based on EUROSTAT 2015 data[42] and FAO Data).

[41] To note that this is in step with estimates in 2013 of area of land being used in Europe for biofuels set out in Ecofys et al (2012) and future estimates by Laborde (2011)
[42] http://appsso.eurostat.ec.europa.eu/nui/show.do?dataset=lan_lcv_ovw&lang=en

	Evidence	Confidence	Calculation Method	Estimates
	Area soybean in Europe – 893 thousand ha – area expanding but bioenergy use of soybean is declining and EU proportion of soy likely to be result of other policy drivers esp. given the rise in production is post 2014 indicating a potential link to measures in the current CAP (greening, VCS). Hence a small proportion of production is considered associated with bioenergy, the assumption here is 5%. Area of sunflower 4,200 thousand ha; area of sunflower production has increased by 19.8% since 2007, corresponding to the RED compliance period. However, it cannot be assumed that all of the rise is driven by bioenergy. Hence 10% of the area is assumed to be dedicated to bioenergy. Cultivated area for total cereals – 57,300 thousand ha. Total production of wheat and spelt in Europe is 48% of total cereal production (Figure 2-6); so proportion of wheat production used for bioenergy is estimated at 4.05% Area of sugar beet production – 1,500 thousand ha; proportion used for bioenergy 12.5% Area of green maize production – 6,200 thousand ha; proportion of maize used for bioenergy 7.5% (proportion for maize not specified in the analysis i.e. all maize or green maize, from associated text this is considered to be all maize). Grain maize accounted for 9,260 thousand ha in 2015. This has been added alongside green maize to the calculation based on evidence from Germany of use of grain maize for biogas.		Area of maize for bioenergy estimated as 1,160 (1000ha)	

	Evidence	Confidence	Calculation Method	Estimates
Conversion of grassland or other semi-natural habitat or fallow to conventional crop	Data exists for conversion of grassland but not for conversion of fallow/semi natural habitats more widely. Long term fallow (economic based on price for commodities) vs fallow in rotation – long term fallow is costly to bring in to production and area of fallow land varies considerably in scale and type across the EU[43]. To note trends of UAA decline generally in response to conversion to urban land uses and expansion of forest lands[44] Based on Eurostat data in 2015 there were 59.6 million ha of permanent grassland in Europe, in 2009 the figure was 62,000 ha. As noted in the EU evaluation of greening measures under the CAP that DE, FR, NL, RO, UK (EN) noted that there is pressure on permanant grassland caused by arable conversion. Whilst ES, LV, UK (Sc) noted little pressure on grassland. Hence the pressure scale and drivers vary across MSs.	Data on permanent grassland used from Eurostat UAA area reporting tables. Noted from the greening evaluation that there are inconsistencies in this data due to the reporting area and change in definitions of permanent grassland in 2015.	2009-2015 loss of 2,470,000 ha of grassland. However, this figure represents total losses. To convert this to losses from arable conversion this figure needs to be adapted. The SOER notes urban land expansion and forest conversion to be the primary loss. Therefore, a conservative estimate of 30% of losses due to arable conversion is applied. This is then adapted considering that in 2015 bioenergy feedstocks from conventional crops accounted for 5.7% of arable land use.	Based on a 30% estimate of attributing conversion of permanent grassland to arable land and of this 5.7% of arable land being attributed to bioenergy feedstocks – conventional crops for bioenergy led to a loss of 42,200 ha of permanent grassland equivalent to 0.07% of permanent grassland in 2015. Equivalent to **0.019% of arable, grassland and shrubland**. To note it is not possible to state which permanent grasslands are being lost i.e. those of high or lower biodiversity importance.
Intensification of grassland management	This occurs at an extremely small spatial scale that cannot be quantified with available data. To note that while biomass is at times used for biogas at	High confidence that it occurs at an extremely small scale.		No data but assumed to be of similar magnitude to Miscanthus on grassland (see

[43] (Allen et al, 2014)
[44] (EEA, 2015a)

	Evidence	Confidence	Calculation Method	Estimates
	present this is often in combination with cutting for feed.			below), i.e. 0.005% of arable, grassland and shrubland
Intensification on existing arable land, mainly due to (conversion to maize)	A switch from cereals and other crops to maize production normally results in intensification. Crop changes for other bioenergy feedstocks normally have relatively neutral biodiversity impacts, or are variable depending on preceding crops. Therefore, this is based only on maize production, using above data	See maize under conventional crops above	See maize under conventional crops above	Area of maize for bioenergy estimated as 1,160,000 ha. In 2015 this represents 1% of arable land in 2015, **and 0.53% of arable, grassland and shrubland**.
Removal of residues from crops (straw etc)	Estimated proportion of residue removals based on JRC 2018 data compared to data for 2015. To note that in turn these are based on modelled exercises – 25% for cereals (57,394,700 ha in 2015); 10% for oil bearing crops (11,555,100 ha in 2015), 10% potatoes (1,656,130 ha), 50% for sugar beet (1420,330 ha). To note that these are extraction rates, it doesn't mean necessarily the rest remains in the soil as farmers will remove to ensure nitrogen balance, enable field working etc Removals are not primarily being used for biofuels, given low penetration of advanced fuels, although some are being used in energy plant (straw) or biogas plant. There is a trend towards higher residue production through expanding areas of higher residue extraction crops i.e. oilseeds. To note crop management decisions and end use decisions are different/lead to different extraction rates at present.	Area figures from Eurostat 2015 data set but estimates of residue extraction are form JRC modelling exercise and data on actual use for bioenergy is limited – although remains relatively low given challenges associated with use of residues for energy and alternative markets. Hence a rate of 10% use of residues extracted is assumed at present for bioenergy.	Total area of key crops calculated, compared to proportion of extraction rate to define an average extraction rate for the area of crops. Crops for which residue calcs exist from JRC are equal to 72,026,260 ha in 2015. On average it was calculated that 22% of residues on this area is being extracted (this is driven by the relatively high extraction rates for cereals and high cereal area). For other crop areas the rate is assumed to	Estimate of current residue extraction rate over total EU arable land is 14.8%. Of this 10% is currently estimated to be used for bioenergy i.e. 1.5%.

	Evidence	Confidence	Calculation Method	Estimates
			be 0% based on JRC data.	
Bioenergy from non-food crops (assumed to be purpose grown for energy) or other feedstocks on grassland and shrubland	Estimated total area of energy crops including Miscanthus and SRC is estimated to be 20,000 ha being used to produce material for bioenergy. This will be split between grassland and arable land. See below.			Total area of energy crops (SRC and Miscanthus etc) estimated at 20,000 ha, or 0.01% of arable, grassland and shrubland. This estimate is split evenly between arable and non-arable as below.
Conversion of grassland and shrubland to Miscanthus etc or SRC	No statistical baseline, but anecdotal evidence that miscanthus is very unlikely to be on this land and is treated primarily as a crop i.e. on existing arable. However, into the future it may be pushed onto other areas given ILUC/extension of agri biomass rules under CAP. Assumed 0 for the baseline analysis. SRC more likely on grassland or marginal land esp. willow. SRC statistics are not formally recorded/consistent at EU level. Based on evidence from the SRCplus project the baseline of 10,000 ha of SRC in 2015 is estimated (consistent with the baseline assumed for RECEBIO I based on modelled and existing data) is used for this calculation. Assumed this is all non-arable land but as SRC is an agricultural crop that this would still be in the UAA i.e. for this calculation assumed to be compared to permanent grassland area as a proxy.	Limited reliable data on area of SRC therefore area is estimated based on a combination of modelling and evidence from case examples. 100% of SRC is assumed to be being used for energy, although it should be noted that this can also be used for material purposes/fibre.	Area of Miscanthus grown on converted grassland – 0ha; areas of SRC grown on converted permanent grassland – 10,000 ha; area of permanent grassland in the UAA assumed to be 47,735,449 ha in 2015 (consistent with baselines used above but noted as inconsistent with Eurostat reporting)	Area of SRC (taken as a proxy for energy crops, see justification) in 2015 is 10,000 ha. **This is 0.0046% of arable, grassland and shrubland.**
Conversion of arable cropland to Miscanthus etc or SRC	Miscanthus is generally treated as an arable crop, it is assumed that current growth is 100% on cropland. Reported data on Miscanthus cropping is inconsistent, therefore a baseline is estimated 15,000 ha is used as the basis of this calculation building estimates from UK and France (who are	Limited reliable data on growth of energy crops across the EU and no consistent figures	Estimated area of Miscanthus (15,000ha), 2/3 of material is estimated to be used for energy i.e.	10,000 ha of arable land devoted to energy crops (principally Miscanthus) in 2015 = 0.014% of arable land. **This is 0.0046% of arable, grassland and shrubland.**

	Evidence	Confidence	Calculation Method	Estimates
	known to be significant growers in Europe). Evidence from France suggests that two thirds of Miscanthus production is currently used for energy, while one third is used for materials including landscape management materials and bedding. Very little evidence of farmers converting arable land to SRC and this will be less likely in future given the emphasis on use of marginal or abandoned land[45]. Assumed 0 for the baseline		approximating to 10,000ha. Comparing Miscanthus area for bioenergy to total area of arable land (107Mha) in 2015.	
Afforestation of grassland and shrubland	FAO data indicate that in 2015 there was 69,000 ha afforestation through planting (i.e. not natural expansion) based on data from 20 Member States (Annex 2). Based on the average % of the expansion in relation to arable, grassland, shrubland and bare land in each country (i.e. 0.05%) it can be estimated that total 2015 afforestation in the EU was 98,000 ha.	Data on forest area is relatively robust although exact data on the locations is limited. It is assumed here that afforestation takes place preferentially on permanent grassland, rather than arable land.	The estimated 98,000 ha planted afforestation is multiplied by 5 to estimate the amount that occurred since 2010, i.e. 490,000. It is assumed that 95% of this was on grassland or shrubland, therefore 466,000. This is multiplied by the 21.9% of forest biomass that is considered to be used for bioenergy (see below).	It is estimated that over 2010-2015 102,000 ha of planted afforestation occurred as a result of bioenergy production. **This is 0.047% of arable, grassland and shrubland.**
Afforestation of arable cropland	Data available from FAO on overall planted afforestation, but this does not indicate previous	Known to occur due to reports of biodiversity	As above with assumption that 5%	It is estimated that over 2010-2015 5,370 ha of planted

[45] See (Allen et al, 2014)

	Evidence	Confidence	Calculation Method	Estimates
	land use. Afforestation assumed to be negligible given the economic consequences of change[46], but does occur on some poor and/or degraded land.	pressures from afforestation, but no relevant data identified – and therefore estimate is imprecise – but high confidence that the area is very small and much lower than afforestation on grassland.	of planted afforestation is on arable land.	afforestation occurred on arable land as a result of bioenergy production. **This is 0.0025% of arable, grassland and shrubland.**
Planting of trees in farmland	This is considered to not contribute significantly in terms of bioenergy. There could be instances of planting trees or other landscape features including windbreaks, field trees, landscape features – but these are likely to be driven by the need to satisfy other goals i.e. GAEC or agri environment schemes. Material from management may be used for bioenergy e.g. you may be required to cut hedgerows but not done because of the energy market although the energy market may provide valorisation for the outputs.			No data but assumed to be similar magnitude to Miscathus on grassland (see below), ie. **0.005% of arable, grassland and shrubland.**
Forest biomass: standing stock (m3) of EU forests used for bioenergy	The total growing stock of EU forests and other wooded land in 2015 was 26,035,561 (1000 m3) (Eurostat) supplied by an area of 160,931 (1000ha). JRC analysis for 2015 estimated that 920,826 (1000m3) was the total wood use in the EU 28 of this 37,588 (1000m3) was net imports meaning that 883,238 was estimated to be domestically produced. Of total usage 451,082 (1000m3) were used for energy of which 429,184 was domestically produced (to note that this production is higher than other analysis). Therefore, 48.59% of domestic	The data on wood use from JRC represents the most up to date and reliable data source. However the JRC analysis of use of primary biomass is not comparable to other sources as the split between direct and indirect wood processing routes does not consider whether materials are extracted	Calculating the wood use for energy domestically produced from primary forest materials in 2015 based on data from JRC 2018 and proportions from Keränen & Alakangas (2014).	In 2010 domestic wood biomass use for energy was 340,917 (1000m3) (based on JRC figures) of this 166,367 (1000m3) are considered to be from primary biomass. This represents 21.9% of total wood use and compared to 0.7% of standing stock in 2010[47]

46 (Alliance Environnement and EFI, 2017)

47 Based on standing stock of 24,132 (Mm3) in 2010 from Forest Europe 2011 for the EU.

	Evidence	Confidence	Calculation Method	Estimates
	wood production was considered to be used for energy. However, this included direct/primary wood sources and industrial by-products and secondary woody biomass. Wood flow analysis (Keränen & Alakangas (2014)) suggests that in 2014 51% of wood material was from primary sources and 48% is from by product flows. This data is considered reliable but applying it hear could lead to some margin of uncertainty given the differing overall volumes considered. However, the 2011 Indufor analysis also identified approximately a 50/50 split in sources.	directly from the forest for processing or are from a secondary source – particularly in the case of charcoal and pellets. Hence the proportion of primary material and secondary material from wood flow analysis are used. This introduces some uncertainty as the proportions will likely increase as wood use profiles change and biomass extraction increases. However, this is the best data available for comparison.	Comparing this to total growing stock in Europe.	In 2015 it is estimated that 429,184 (1000m3) wood was used for energy, of this 218,883 (1000m3) were considered to be energy from primary forest biomass i.e. extracted directly from the forest (as both residues and roundwood) for energy use. **This is 24.8% of the total wood use in Europe in 2015** and represents 0.8% of the total standing stock of European forests in 2015.

Figure 2-26 Estimated use of bioenergy feedstocks to 2030 (based on baselines set out in RED II)

Estimated change based on analysis of future estimates compiled in sections 2.1 to 2.3 and cited studies cited in these sections and/or in footnotes. Estimates are only made for categories for which a baseline estimate was made in Figure 2-25. To note – Annex 2 includes extended text which explain sources.

NB. Text and figures in bold indicate assumptions and estimates that are used in the semi-quantitative vulnerability assessment in chapter 4.

	Estimates for 2015	Evidence of change 2030	Confidence	Calculation Method	Proportional change to 2030
Conventional crops (i.e. food and feed/forage crops) going to biofuels	Land utilised to cultivate conventional feedstocks in Europe – 6.05 million ha; approximately **3.6% of 166 million ha arable land and**	The following estimates are based on data from the European Commission's Agricultural Outlook 2018-2030. Up to 2030 use of conventional crops for biofuel production will decline but only by a limited percentage i.e. less than 5 per cent. For the purposes of providing future projections in	It should be noted that this is expressed in this study as a decline in area used for these feedstock crops, however, over the same period increases in yield will also influence how this	Area of rapeseed for bioenergy estimated as – 3138 (1000ha) in 2015, decline by 2% = 3075 (1000ha) Area of soybean for bioenergy estimated as – 45	No proportional change is expected, so land utilised to cultivate conventional feedstocks in Europe – approximately **2.8% of**

	Estimates for 2015	Evidence of change 2030	Confidence	Calculation Method	Proportional change to 2030
	grassland in 2015[48], and 5.7% of arable land (based on 107 million ha).	the context of this study the figure of 2% decline is assumed for biodiesel feedstocks (rapeseed, soybean, sunflower), 4% for bioethanol feedstocks (wheat). Biogas agricultural crops will also be subject to restrictions from 2020 under RED II i.e. in terms of the criteria applied to sourcing of agricultural material. Rules have been adopted in a number of Member States restricting the use of primary crops being used as feedstocks for biogas facilities and promoting the use of manure and residue-based systems. Non the less, given wider promotion of biogas, AEBIOM analysis suggests that there is considerable additional biogas capacity needed to meet Member States 2020 NREAP targets. In 2014 capacity was 15.612ktoe with the EU28's NREAPs stating 20.820ktoe is needed to 2020. This implies a continued expansion primarily of agricultural based biogas production. Therefore to 2030 it is assumed that biogas based on conventional crops will expand. This is expressed as a 10% increase in the use of maize for biogas production, although this potentially could be met by other materials, residues or alternative crops.	translates into actual production and land management changes. Moreover, as noted for rape and sunflowers the proportion and distribution between the feedstocks will alter over this period as prices fluctuate and alternative products emerge. The locations in which core crops are grown would also likely fluctuate. Biogas expansion and use of conventional crops is anticipated to continue to increase albeit at a reduced rate to 2030. However, most existing plants have the right to retain their feedstock sourcing for the duration of existing contracts (hence no decline) and crop-based biomass remains a key source of feedstocks, even if combined with other residual sources.	(1000ha), decline by 2% = 44 (1000ha) Area of sunflower for bioenergy estimated as – 420 (1000ha) decline by 2% = 412 (1000ha) Area of wheat for bioenergy estimated as – 1115 (1000ha) decline by 4% = 1071(1000ha) Area of sugar beet for bioenergy estimated as – 179 (1000ha) decline by 4% = 172 (1000ha) Area of maize for bioenergy estimated as 1,158 (1000ha) increase by 10% = 1274 (1000ha)	**arable, grassland and shrubland**
Conversion of grassland or other semi-natural habitat	Based 30% estimate of attributing conversion of permanent grassland to	The estimates suggest a relatively constant level of total conventional crop area vs arable or agricultural land being used for			To note that the proportion of arable land and UAA used for

48 To note that this is in step with estimates in 2013 of area of land being used in Europe for biofuels set out in Ecofys et al (2012) and future estimates by Laborde (2011)

	Estimates for 2015	Evidence of change 2030	Confidence	Calculation Method	Proportional change to 2030
or fallow to conventional crop	arable land and of this 5.7% of arable land being attributed to bioenergy feedstocks – conventional crops for bioenergy led to a loss of 42,200ha of permanent grassland equivalent to 0.07% of permanent grassland in 2015. This compares to an estimated 95,059 (1000ha) of grassland and shrubland in the EU in 2009 (i.e. a loss of 0.04 of the total) based on FAO stat figures. It should be noted between 2009 and 2015 total grassland and shrubland in Europe increased in area based in FAOstat data, while area of permanent grassland declined.	bioenergy between 2015 and 2030. However, this potentially masks increase in likely biogas feedstocks, specifically maize in the calculation. In contrast there are anticipated declines in other feedstocks. It is likely, therefore, that localised pressures on permanent grasslands associated with conventional crops for bioenergy remain, despite tempering of expansion of biofuels from food and feed crops.			conventional crop-based feedstocks is considered to remain relatively static to 2030.
Removal of residues from crops (straw etc)	Estimate of current residue extraction rate over total EU arable land is 14.8%. Of this 10% is currently estimated to be used for bioenergy i.e. 1.5%	Agricultural residues are seen as a high potential opportunity for biomass for energy that can deliver against GHG emission saving goals and wider sustainability needs. Moreover, their use is actively promoted in the REDII rules. Estimates[49] suggest that a third of residues from agriculture should be available for energy use – based on leaving a third available to retain soil quality and a third for	There are no baselines set for competing uses of residues or levels of residues to remain in soils. Hence the comparators may be unrealistic, however, these are the norms used in the literature	20% considered unavailable (see evidence column) and 66% to remain in soils or to be allocated to non-energy uses leaving 14% of agricultural residues for bioenergy uses	It is estimated that 14% of residues from agriculture could be used in energy in 2030 representing a significant increase from 1.5% in 2015. An increase of over 900%

[49] from (Malins et al, 2013)

	Estimates for 2015	Evidence of change 2030	Confidence	Calculation Method	Proportional change to 2030
		other uses. However, 100% of residue utilisation is also considered ambitious given challenges associated with collection and distribution. Hence 10% of residues are considered to remain beyond extraction. Moreover, certain crops are unlikely to be available as residues (for example legumes are part of a management cycle for N and as such are ploughed in to fields) hence an additional 10% is deducted from the calculation for crops not considered to be residue appropriated.			between 2015 and 2030.
Bioenergy from non-food crops (assumed to be purpose grown for energy – terminology for space for energy report) or other feedstocks on agricultural land					The combined area for energy crops i.e. Miscanthus and SRC in 2030 is estimated to be 5.3 Mha dedicated to meeting bioenergy demand. **This is equivalent to 2.4% of arable, grassland and shrubland**.
Conversion of grassland or other semi-natural habitat to Miscanthus etc or SRC	Area of SRC (taken as a proxy for energy crops, see justification) in 2015 is 10,000 ha compared to area of permanent grassland in 2015 is 0.02% or 0.01 of total grassland and shrubland in the EU in 2015 (FAOstat)	The RECEBIO I analysis identified an increase in SRC under its BAU scenario from 10,000 ha in 2010, to 2,500,000 ha by 2030 and to 3,400,000 ha in 2050 and under its EU Emission Reduction Scenario (where ambition and use of bioenergy increases to 2050) to 8,900,000 ha in 2050. Under Recebio I SRC is seen to expand at the expense primarily of the 'other natural land category'. Under Recebio II analysis SRC expanded even further to 15,000,000 ha in	Estimates of SRC expansion are known to by high compared to historic levels of penetration. However, in the absence of SRC expansion sensitivity analysis suggests additional afforestation and other changes. Hence as a representation of change this is felt important to	100% of SRC is assumed to be used for bioenergy in 2030 and 2.5Mha.	Area of SRC in 2030 (taken as a proxy for energy crops) = 2,500,000 ha. Compared to area of grassland and shrubland in 2015 this represents is 2.6%. **This is equivalent to 1.1% of arable, grassland and shrubland**.

	Estimates for 2015	Evidence of change 2030	Confidence	Calculation Method	Proportional change to 2030
		2050. However, given concerns regarding whether SRC transition can be delivered more conservative numbers have been used here. We therefore take a conservative estimate (based on RECEBIO I) of SRC expansion to 2.5 Mha in 2030.	represent here. However, given concerns re likely expansion relatively conservative estimates of SRC expansion from RECEBIO I have been used as a baseline and they have not been adapted to take account of the increased demand under REDII for biomass anticipated to deliver the 32% target.		
Conversion of arable cropland to Miscanthus etc or SRC	Proportion of arable land devoted to Miscanthus (taken as a proxy for energy crops) in 2015 0.014% (10,000ha)	As noted in modelling exercises analysing biofuel production and biomass for heat and power energy crops including Miscanthus are anticipated to expand. Here Miscanthus expansion is used as a proxy for wider energy crops, which are identified in modelling studies as expanding on arable land rather than on other land/grassland (although there may in indirect impacts associated). Utilising the same assumptions as for the SRC analysis a relatively conservative assessment basis for energy crops/Miscanthus expansion is adopted. This is based on RECEBIO I estimates to 2030 and is not adapted to increase estimates to a 32% (compared to a 27%) renewable energy target. This is given the relatively slow expansion see in such crops to date. Under RECEBIO I lignocellulosic energy crops were anticipated to expand to 2.8Mha by 2030.	Estimates for energy crop expansion are known to be high compared to historic adoption patterns. See notes above	Expansion in Miscanthus area (taken as a proxy for lignocellulosic energy crops) in 2030 to 2.8Mha	Proportion of arable land devoted to Miscanthus (taken as a proxy for energy crops) in 2030 is 2,800,000 ha. **This is 1.3% of arable, grassland and shrubland**.

	Estimates for 2015	Evidence of change 2030	Confidence	Calculation Method	Proportional change to 2030
Afforestation of grassland or other semi-natural habitat	It is estimated that over 2010-2015 102,000 ha of planted afforestation occurred as a result of bioenergy production. This is an average rate of 20,400 ha per year.	Expansion in forest area is noted as a trend by the EEA[50] and in data for the EU 28 recorded by Forest Europe. In 2015 there was 160,931,000 ha of forest in the EU 28. Between 2010 and 2015 this increased by 1,695,000 ha i.e. an annual average of 339,000 ha per year). Most of this will be natural expansion and part will be active afforestation for bioenergy and other uses. If the baseline average afforestation rate for the 2010 to 2015 period is taken (i.e. 339, 000 ha per annum) between 2015 and 2030 this would account for an expansion in forest area of 5,085,000 ha. RECEBIO I estimated an increase in EU forest area of approximately 9,000,000 ha between 2010 and 2030 based on the BAU scenario (a rate of increase of 450,000 ha per annum on average over the 20 years). RECEBIO II estimated approximately 10,000,000 ha of afforestation over the same period under its BAU analysis (i.e. a rate of 500,000 ha per yr). Extrapolating the 2010-2015 afforestation rates from 2010 and 2030 would result in 6,780,000 ha of afforestation between 2010 and 2030. Given wider pressures driving demand for wood and forests across Europe it has been decided to adopt the higher afforestation rate based on the RECEBIO I figures.	Afforestation will occur because of a number of pressures and shifts in support and economic viability of forestry. Moreover, there are Member States actively promoting forest expansion but not specifically for bioenergy but for carbon sequestration and wider bioeconomy delivery goals. The RECEBIO estimates consider total wood use so only a portion of the demand anticipated would be attributable to bioenergy specifically, as part of wider shifts in demand for woody biomass for material uses.	It is not possible to use the RECEBIO afforestation rate directly, as the 2015 baseline used in this study was based on actual observed FAO planted afforestation data. But the 147% proportional increase from 339,000 ha per year to 500,000 ha is used to adjust the 2015 estimate. The afforestation is anticipated to occur on grasslands and semi-natural habitats rather than on arable land. This is consistent with known trends to date and land use-based scenario modelling.	Based on the baseline 2015 afforestation rate driven by bioenergy of 20,400 ha per year and the assumed 147% increase, this will lead to 450,000 ha of afforestation for bioenergy by 2030. **This is 0.21% of arable, grassland and shrubland**

	Estimates for 2015	Evidence of change 2030	Confidence	Calculation Method	Proportional change to 2030
		It should be noted both in RECEBIO I and II afforestation came at the expense of the conversion of other natural land categories within the study i.e. predominantly semi natural habitats.			
Afforestation of arable land	It is estimated that over 2010-2015 5,370 ha of planted afforestation occurred on arable land as a result of bioenergy production. This is a rate of 1,074 ha per year.	See above on afforestation of grassland or other semi-natural habitat	See above on afforestation of grassland or other semi-natural habitat	As above, based on 147% increase in annual rate of afforestation, with assumption that 5% of planted afforestation is on arable land.	Based on the baseline 2015 afforestation rate driven by bioenergy of 1,074 ha per year and the assumed 147% increase, this will lead to 23,700 ha of afforestation for bioenergy by 2030. **This is 0.011% of arable, grassland and shrubland.**
Forest biomass: of standing stock (m3) of EU forests used for bioenergy	In 2015 it is estimated that 429,184 (1000m3) wood was used for energy, of this 218,883 (1000m3) were considered to be energy from primary forest biomass i.e. extracted directly from the forest (as both residues and roundwood) for energy use. This is 24.8% of the total wood use in Europe in 2015 and represents 0.8% of the total standing stock of European forests in 2015.	A key challenge for estimating to 2030 wood use is that estimates of total wood use in the modelling and calculation estimates are not equivalent. However, what are considered to be consistent and comparable are trends seen in the real-world data from JRC 2018 and modelled data on increasing proportion of wood biomass being used for energy. The modelling analysis, however, tends to underestimate the level of increase that has already occurred to 2015 based on the JRC figures. If you take the average rate of increase between 2010 and 2015 to estimate potential wood use to 2030, the average rate of increase in wood use	As demonstrated in the evidence column, estimates in terms of future wood demand are based on significantly lower baselines than those being reported in recent JRC analysis for 2015. Hence assessments here should be considered very much as estimates to provide context in terms of demand.	Total wood consumption in 2030 is estimated to be wood consumption of 1,095,000,000m3 in 2030. Modelling analysis under RECEBIO I also identified a change in the proportions of wood being used for energy and the use of primary vs secondary biomass as wood for energy increased in prevalence. Hence in 2030 approximately 54% of woody biomass is being used for energy with 55% estimated	In 2030 it is therefore estimated that 325,215 (1000m3) of primary wood is being used for energy (an increase of approximately 48% in terms of total wood use for energy compared to 2015). **This is 29.7% of the total wood use in Europe in 2030** and would be equivalent to 1.2% of standing stock in 2015.

	Estimates for 2015	Evidence of change 2030	Confidence	Calculation Method	Proportional change to 2030
		between 2010 and 2015 is 36,700,000m3 per year. Multiplied by 15 this would give an additional demand of 550,500,000m3 between 2015 and 2030 resulting in a total wood use in 2030 of 1,471,000,000m3 significantly exceeding estimates under both RECEBIO I and II for 2030 and 2050. If only the expansion for bioenergy is taken into account between 2010 and 2015 this provides a rate of 18,600,000m3 per year. This compares to the estimated increase based under the BAU scenario of RECEBIO I using 163,000,000m3 in total between 2010 and 2030 or approximately 8,000m3 per year. Given these estimates are so vastly different it has been decided to estimate increase based on proportional volume change under RECEBIO I rather than totals (to take account of differing baselines). RECEBIO I saw an increase of 19% in wood use to 2030. If applied to the 2015 data this would provide for an increase of 174,956, 000m3 from the 2015 baseline arriving at a total wood consumption of 1,095,000,000m3 in 2030. This is considered to be achievable and is within the bounds of estimate of potential wood extraction in the RECEBIO studies to 2050.		from primary biomass sources.	

3 Assessment of the sensitivity of EU protected habitats and species to bioenergy feedstock production

3.1 Methodology and evidence sources

This element of the study step narrowed down the scope of the habitats and species to be considered in the sensitivity assessments by identifying those that are most likely to be affected by bioenergy production. From previous assessments (see later discussion), it is clear that the specific habitats and species most likely to be impacted by bioenergy feedstock production are those of agricultural habitats, grasslands, forests, and heath or shrub land; and therefore these were the focus of this sensitivity assessment and the vulnerability assessment, as carried out in chapter 4.

Bioenergy production also has significant impacts on aquatic habitats, but these are primarily indirect and are very difficult to assess and quantify. It was therefore not feasible to assess them in this study, but the likely impacts on aquatic habitats and species should be borne in mind (Box 3.1).

Box 3.1 Bioenergy impacts on aquatic habitats and species

Bioenergy production has a range of impacts on aquatic habitats and species. The most detrimental impact is through the conversion of permanent grassland into arable crops, which leads to an increase in soil run-off and a release of nitrogen and phosphorus from the soil into watercourses. For example, maize crops cause significantly increased soil erosion and sedimentation compared to grassland, as well as much higher levels of nutrient runoff and leaching to surface and groundwater, as it is a tall crop with widely spaced stalks, leaving relatively large areas of soil exposed throughout the growing season. The crop reduces water quality in surface waters compared to grassland. For example, a survey of arable crops in England showed that 75% of the late harvested maize sites had high or severe levels of soil degradation that generated enhanced surface-water runoff (Palmer and Smith, 2013). This has a negative effect on freshwater habitats and species. Sugarbeet and sunflowers also have a high proportion of exposed soil and relatively slow growth, and are therefore prone to high soil erosion rates. The planting of biomass crops such as Miscanthus and short rotation coppice on grassland sites will also result in soil erosion compared to the grassland, although the impacts will decline once the crops develop.

Some bioenergy crops could reduce run-off if they are grown on existing arable land, and therefore provide benefits for aquatic ecosystems. For example, Miscanthus and short rotation coppice provide more soil cover than annual arable crops, and because they remain in the field over several years without significant soil disturbance, their overall impact on aquatic ecosystems is much lower than arable crops. The crops may additionally improve soil organic matter content and thereby decrease soil erosion in subsequent crops, provided the ground cover is not controlled intensively. However, it is generally likely that most dedicated bioenergy crops will be grown on grasslands and therefore provide little if any additional benefits in terms of water quality.

The case of Latvia is especially striking as it reported freshwater and wetland HD habitats as under high pressure from forest clearance over the 2007-2012 reporting period.

For each of the three bioenergy feedstock types, the sensitivity assessment entailed a review of two key sources of evidence as described below.

Firstly, Member State reporting data on pressures on EU protected habitats and species submitted in accordance with the requirements of the Birds Directive (under Article 12) and

Habitats Directive (Article 17) were analysed (Box 3.2)[51]. The aim was to identify EU protected species and habitats subjected to pressures associated with bioenergy feedstock production activities and list them in tables in relation to each biofuel feedstock type.

The data were extracted from the Article 17[52] and Article 12[53] databases held by the EEA and form the source for all the pressure analysis tables below, unless otherwise indicated. It should be noted that the use of these databases has some limitations. For instance, the pressures reported in these databases are divided according to a hierarchical classification system whereby Member States add information at the level they consider most suitable following some general recommendations (Evans and Arvela, 2011; N2K Group, 2011). This leads to differences in the level of precision that prevent the information contained in the databases from being fully standardised. In this study we used the 3rd (and sometimes 4th) hierarchical level in order to have the necessary detail on the pressure, but we note that some Member States did not report to this level of detail for some habitats and species[54].

Box 3.2. The information provided by Member States on pressures and threats on EU protected habitats and species in accordance with the requirements of the Birds Directive and Habitats Directive.

Standardised reporting forms and guidelines have been produced for Member States to report as required under Article 12 of the Birds Directive (N2K Group, 2011) and Article 17 Habitats Directive (Evans and Arvela, 2011). According to these, over the 2007-2012 reporting period for birds, and 2007-2012 period for HD habitats and species, Member States were required to identify and rank pressures affecting each habitat and species, using a common checklist of codes[55]. Pressures are considered to be factors that are acting now or which were acting during the reporting period, whilst threats are expected in the future. Under the Habitats Directive, reporting threats are distinguished from pressures and reported separately, but no distinction is made for birds. Up to 20 pressures / threats can be listed, but Member States are recommended to give as short a list as possible. The checklist in the reporting guidance groups pressures / threats under 17 headings and has 75 categories at the 2nd hierarchical level (e.g. A01 cultivation). There is the option to provide more detailed information through 3rd and 4th level codes.

The relative importance of each of the pressures / threats entered must be ranked in one of three categories:

H = High importance/impact: Important direct or immediate influence, and/or acting over large areas.

M = Medium importance/impact: Medium direct or immediate influence, mainly indirect influence, and/or acting over moderate part of the area/ regionally only.

L = Low importance/impact: Low direct or immediate influence, indirect influence, and/or acting over small part of the area/ locally only.

To help identify the most important factors at an EU scale, the number of entries with the highest rank (H) is limited to a maximum of 5 data entries.

For HD habitats and species, reporting is carried out in relation to biogeographic zones[56]. The codes for these used in the tables below are:

[51] This report was written between March 2018 and February 2019; therefore it was not possible to use the Member States Article 17 reports for the 2012 to 2018 period.

[52] https://www.eea.europa.eu/data-and-maps/data/article-17-database-habitats-directive-92-43-eec-1

[53] https://www.eea.europa.eu/data-and-maps/data/article-12-database-birds-directive-2009-147-ec/article-12-data

[54] e.g. Sweden, Lithuania and the UK did not report at 3rd and 4th levels for their forest habitats, whilst reporting from Denmark and the Czech Republic on their forest habitats was partly at 2nd level and partly at 3rd level, as was reporting from Germany and Spain on their grassland habitats

[55] Available at the Article 12 and Article 17 reference portals https://bd.eionet.europa.eu/activities/Reporting/Article_12/reference_portal

[56] https://www.eea.europa.eu/data-and-maps/figures/biogeographical-regions-in-europe-2

Alpine (ALP)
Atlantic (ATL)
Black Sea (BLS)
Boreal (BOR)
Continental (CON)
Macaronesia (MAC)
Mediterranean (MED)
Pannonian (PAN)
Steppic (STE)

Member States are required to indicate the method they used to identify the pressures and their severity, with the evidence base and method either 1) expert opinion, 2) mainly based on expert judgement and other data, or 3) based exclusively or to a larger extent on real data from sites/occurrences or other data sources. It was not practical to sub-divide the analysis according to this additional level of information in the habitat and species assessments, but interpretation of the results should take into account the general level of reliability of the information. Of the 7,350 species assessments in 2007-2012, 16% used the most rigorous method 3), 50% the semi-rigorous method 2), and 26% only expert opinion (method 1). 8% did not report the method used. Of the 3,117 habitat assessments, 18% used the most rigorous method 3), 60% the semi-rigorous method 2), and 19% only expert opinion (method 1). 3% did not report the method used.

All species and habitats reported as being subject to low, medium or high pressures by biofuel production (A06.03) were identified and listed as the pressure category is very specific to bioenergy production. No other pressure types in the reporting system are as specific to bioenergy. However, the following pressure types may, in some circumstances be the result of bioenergy feedstock production on agricultural land:

- grassland removal for arable land (A02.03),
- artificial planting on open ground (non-native trees) (B01.02).

The following pressure types may in some circumstances be the result of use of forest wood or residues for bioenergy:

- removal of dead and dying trees (B02.04),
- removal of forest undergrowth (B02.03),
- thinning of tree layer (B02.06),
- forest clearance (B02.02),
- forest exploitation without replanting or natural regrowth (B03),
- forest replanting with non-native trees (B02.01.02),
- use of biocides, hormones, and chemicals in forestry (B04),
- use of fertilizers (forestry) (B05).

The habitats and species reported as under high level pressure from these categories are listed in the relevant sections of this chapter. Those species most frequently reported as subject to the above-mentioned pressures are listed in Annex 2.

Secondly, a review was carried out of the scientific literature on the direct and indirect effects of the production of the three main types of bioenergy feedstock[57] on habitats and species to identify sensitive EU protected habitats and species, as well as the causes of this sensitivity, i.e. risk factors. This involved: a) defining the required scope of the literature search and classification criteria (feedstocks, impact types etc) and biodiversity components; b)

[57] I.e. Conventional crops (i.e. food and feed/forage crops), energy crops on agricultural land, and forest biomass from existing forest land.

identification of relevant literature using web searches and examination of reference lists in identified relevant literature; c) critical appraisal of the identified literature for reliability (e.g. taking into account research design, sample sizes and age) and relevance (e.g. with respect to location and feedstock type) to draw conclusions on the sensitivity of the related habitats and species and feedstock types. To assist with this, a literature database (in Excel) was compiled with search terms and other keywords, and a summary of evidence of each impact, and the relevant biodiversity component(s) and bioenergy type(s) for each reference.

This study initially considered the principal existing studies of the overall impacts of bioenergy on biodiversity. This included a previous review of policies that could mitigate the impacts of bioenergy on biodiversity, which pointed out that the literature was primarily retrospective, focused on few species, countries (mainly in the US and Canada), and ecoregions, and fraught with generalisations from weak inference (Northrup and Wittemyer, 2012). Similarly, a review of published impacts of bioenergy crop production on biodiversity found relatively little evidence from the EU, concluding that reported impacts depend on initial land use, and that the impacts of second generation bioenergy crops tend to be less negative than first generation ones, and are in some cases positive (at the field level), in particular in temperate regions (Immerzeel et al, 2014). A recent review of the impacts of bioenergy in the Mediterranean region cited only one case of impacts on biodiversity (Pulighe et al, 2019), namely on carabid beetle communities which are not directly protected by the EU Nature Directives.

A study by Louette et al (2010) used the BIOSCORE database of the habitat requirements of selected key species, and land use models, to assess the potential impacts of three biofuel policy scenarios on biodiversity (Box 3.3). The study only focused on the policy scenario of large-scale second-generation bioenergy crops cultivation throughout Europe (i.e. woody species, particularly willow *Salix* spp. and poplar *Populus* spp., which are harvested over a short rotation cycle of about 10 years). Under this scenario, it is assumed that land use change will mainly arise from the conversion of open agricultural land and abandoned land that will be taken back into production. Loss of existing forest is foreseen to be minimal in this context. The results indicated that the possible impacts vary spatially and depend on the biofuel crop choice, with woody crops being less detrimental to biodiversity than arable crops. Large-scale short-rotation coppice production on open agricultural land and abandoned land would have a negative net effect on all taxonomic groups except plants. The model predicted negative effects of the second-generation bioenergy crop scenario for 40% of reptiles in all regions, around 25% of butterflies in the Atlantic, Continental and Mediterranean regions, and 25% of species in the Boreal region. It also predicted that more than 35% of bird species in the Mediterranean region would be negatively affected.

As these previous reviews, and the modelling study, provided only broad insights that related to biodiversity in general, they are of limited relevance to this study of impacts on EU protected habitats and species. This study has therefore primarily drawn on a more targeted review of evidence of impacts of all the main types of bioenergy on the EU protected habitats and species.

Box 3.3. Assessment of possible impacts of biofuel policy on European biodiversity using the BIOSCORE model

The BIOSCORE model was built to allow the assessment of possible impacts of changing environmental variables on European biodiversity (Delbaere et al, 2009). The model uses the habitat preferences of a large set of key species at the European scale (Louette et al, 2010). It uses land use projections for 17 categories of land use including arable and woody biofuel crops, based on an aggregation of the CORINE land cover data (Eggers et al, 2009). BIOSCORE uses a combination of different models to spatially allocate the different land uses, including a global general equilibrium model, an integrated assessment model that accounts for environmental and land-based impacts, and the Dyna-CLUE model.

BIOSCORE was used to assess the potential impacts of land-use changes resulting from a change in the current biofuel policy on biodiversity in Europe (Eggers et al, 2009; Louette et al, 2010). Three different policy options were explored based on a land use scenario for 27 EU countries for the period 2000 to 2030 at 1km resolution: policy option (e1) of no or low ambition on biofuels (0% blending obligation on share of biofuels in transport sector in 2010 and kept constant afterwards); policy option (e2) of medium ambition on biofuels (5.75% blending obligation on share of biofuels in transport sector in 2010 and kept constant afterwards); policy option (e3) of high ambition on biofuels (11.5% blending obligation on share of biofuels in transport sector in 2010 and kept constant afterwards). Policy option (e2) was used as a reference scenario, as it reflected the target of the EU Renewable Energy Directive (RED) in 2010.

The BIOSCORE model has since been updated, but the updated version has not been applied to the biofuel scenarios (Hendriks et al, 2016; Vermaat et al, 2016).

3.2 Biodiversity impacts of bioenergy from conventional food crops – maize, oilseed rape, cereals, grass

3.2.1 Introduction

This section assesses the following bioenergy feedstock types:

a. Crops for biofuel or biogas:
   i. Oilseed rape
   ii. Maize
   iii. Wheat or other cereals
   iv. Sugar-beet
   v. Grass
   vi. Others

The production of bioenergy from conventional crops may lead to the following direct and indirect risk factors for EU protected species and habitats:

- Conversion of grassland or other semi-natural habitat or fallow to cropland.
- Intensification of grassland management (e.g. higher fertiliser use, re-seeding of grassland, high stocking rates and/or switch to silage).
- Crop changes on existing arable land (mainly cereals to oil-seed rape, and arable crops to maize) and associated intensification of cropland management (e.g. higher fertiliser & pesticide use).
- Removal of residues from crops (straw etc).

These risk factors and the evidence for their impacts on EU protected habitats and species are described below.

3.2.2 Conversion of grassland, fallow or other habitats to cropland

General biodiversity impacts

The conversion of grasslands to conventional crops for bioenergy (or as an indirect effect of bioenergy expansion through Indirect Land Use Changes - ILUC), results in such profound biophysical and vegetation changes that the converted habitat can be considered to be destroyed together with the vast majority of most of its associated species. This results in major biodiversity losses, especially if it affects semi-natural habitats. This is because it is clear from numerous studies (e.g. reviewed in Poláková et al, 2011; Stoate et al, 2009) that the key determinant of the richness and abundance of biodiversity associated with agricultural habitats is the degree to which they have been modified from their natural state as a result of agricultural improvements (e.g. draining, ploughing and reseeding of grass, conversion from grasslands to crops) and the intensification or modernisation of management (e.g. cultivations, the use of fertilisers, irrigation and pesticides) and specialisation in particular intensive systems. As a result, croplands are hostile to many species and often no longer provide sufficient food resources for species that would otherwise tolerate the conditions. In contrast, semi-natural agricultural habitats are of particular value for rare and otherwise threatened species of open habitats because they provide grass and shrub dominated habitats that are similar to previously present natural ecosystems (such as steppic grasslands) and provide the species' specialised ecological requirements.

Consequently, most remaining semi-natural agricultural habitats in the EU are of very high conservation value as they are much diminished, and many support species that are restricted to such habitats, many of which are also rare or endemic. As a result, most natural and semi-natural agricultural habitats in the EU are listed on Annex I of the Habitats Directive, and many associated species are listed in Annex I of the Birds Directive and Annex II of the Habitats Directive, as they are also highly or exclusively dependent on natural or semi-natural habitats (Figure 3-1).

The biodiversity impacts of converting agriculturally improved grasslands to croplands are much lower, as such grasslands are at least an order of magnitude lower in their biodiversity value than semi-natural grasslands, and consequently do not include any Annex I habitats as a result of the impacts of drainage, fertiliser use, and reseeding (Poláková et al, 2011). In fact, silage fields are often sown grass monocultures with no higher plants of high conservation value present at all, and therefore also have a highly impoverished fauna. Conversion of such grasslands to some crops, such as oil-seed rape, may provide some biodiversity benefits, such as breeding sites for some birds, and flower resources for some pollinators (Riedinger et al, 2015), but these are unlikely to relate to any EU protected species.

Figure 3-1 Agricultural habitats in the EU, their importance for selected threatened habitats and species, and their overall biodiversity

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

Source: (Poláková et al, 2011). 1 (Halada et al, 2011); 2 adapted from (Tucker and Evans, 1997); 3 adapted from (Van Swaay, Warren and Lois, 2006) using updated annexes available from Butterfly Conservation Europe (http://www.bc-europe.org/upload/Butterfly%20habitats%20-%20Appendix%201.pdf): 4 (Temple and Cox, 2009); 5 (Cox and Temple, 2009).
Note: Habitat divisions for each taxa group reflect the habitat types distinguished in the available data.

Bioenergy crops may also be grown on arable land that has been fallow for many years, and this can result in significant biodiversity impacts, as long term fallow can provide habitat for many species associated with farmland, e.g. for ground-nesting farmland birds, small mammals, arable plants and some pollinators (Tscharntke, Batáry and Dormann, 2011; Van Buskirk and Willi, 2004). Long-term fallow supports EU protected species in some areas of extensively managed dry cereals (Figure 3-1). These habitats have sparse crops, high crop rotation diversity and retain a sizeable proportion of fallow and the presence of patches of semi-natural vegetation (Bota et al, 2005; Suárez, Naveso and de Juana, 1997). Such extensive cropping systems are rare, but they occur in parts of eastern and southern Europe, and hold relatively species-rich plant and invertebrate communities, which in turn support abundant small mammal and bird populations including HD and BD species (described below).

Conversion of improved permanent grassland to energy crops also causes indirect impacts on protected habitats and species due to the negative environmental effects. Maize crops cause significantly increased soil erosion and sedimentation compared to grassland, as well as much higher levels of nutrient runoff and leaching to surface and ground water, as it is a tall crop with widely spaced stalks, leaving relatively large areas of soil exposed throughout the growing season. The crop reduces water quality in surface waters compared to grassland. For example, a survey of arable crops in England showed that 75% of the late harvested maize sites had high or severe levels of soil degradation that generated enhanced surface-water runoff (Palmer and Smith, 2013). This has a negative effect on freshwater habitats and species. Sugarbeet and sunflowers also have a high proportion of exposed soil and relatively slow growth and are therefore prone to high soil erosion rates.

Evidence of impacts of habitat conversion to arable on HD habitats

Member States reported under the Habitats Directive that **grassland conversion to arable** is causing the loss of areas of HD grassland habitat types in nine Member States (Figure 3-2). Biofuel production was reported as a low pressure on hay meadows in Germany and subcontinental peri-Pannonic scrub in the Pannonian region of Romania (Figure 3-3). Austria, Belgium, Bulgaria, Ireland and Slovenia reported grassland loss to arable as a high-level pressure on hay meadows. Slovakia, Bulgaria, Cyprus, Belgium, and Spain reported it as a pressure on other grassland habitats. However, these Member States did not report biofuel production specifically as a pressure on these habitats. This, and the fact that only two Member States in total listed biofuel production as a pressure (as low intensity pressure), suggests that biofuel production is not a major direct driver of grassland conversion to arable. Instead it seems to be primarily due to conversions for other purposes, including food and fodder, but also potentially for maize production for bioenergy. However, care should be taken with this interpretation because it is difficult to ascertain the drivers of crop changes, and therefore Member States may have chosen to list the more general pressure code of arable conversion rather than attribute it to a particular cause, thereby also capturing the pressure of indirect land use change. Furthermore, as indicated below in relation to specific Member States, there are other sources of evidence from the literature that suggest that conversion for bioenergy is a significant pressure, especially in relation to maize production.

Figure 3-2 Habitat types reported by Member States as under high pressure from conversion of grassland to arable in the 2007 to 2012 reporting period

Key to biogeographical regions: ALP Alpine; ATL Atlantic; BLS Black Sea; BOR Boreal; CON Continental; MAC Macaronesian; MED Mediterranean; PAN Pannonian; STE Steppic.

Habitat code	Habitat name	MS reporting high pressure of grassland conversion to arable
6440	Alluvial meadows of river valleys of the Cnidion dubii	BG (CON)
6430	Hydrophilous tall herb fringe communities of plains and of the montane to alpine levels	BG (ALP, BLS, CON)
6510	Lowland hay meadows (*Alopecurus pratensis, Sanguisorba officinalis*)	AT (ALP & CON), BE (ATL & CON), BG (CON), DE (ATL & CON), IE (ATL), SI (ALP & CON)
6420	Mediterranean tall humid grasslands of the Molinio-Holoschoenion	CY (MED)
6410	Molinia meadows on calcareous, peaty or clayey-silt-laden soils (Molinion caeruleae)	BE (ATL & CON), BG (ALP)
6520	Mountain hay meadows	BG (ALP, CON)
6250	Pannonic loess steppic grasslands	SK (PAN)
6220	Pseudo-steppe with grasses and annuals of the Thero-Brachypodietea	ES (ATL)
6230	Species-rich *Nardus* grasslands, on siliceous substrates in mountain areas (and submountain areas, in Continental Europe)	BE (ATL)
3260	Water courses of plain to montane levels with the Ranunculion fluitantis and Callitricho-Batrachion vegetation	DE (ATL)

Figure 3-3 Habitat types reported by Member States as under pressure from biofuel production in the 2007 to 2012 reporting period

Key to biogeographical regions: ALP Alpine; ATL Atlantic; BLS Black Sea; BOR Boreal; CON Continental; MAC Macaronesian; MED Mediterranean; PAN Pannonian; STE Steppic.

Habitat code	Habitat name	Intensity of pressure from biofuel production			Evidence in the literature
		high	medium	low	
6520	Mountain hay meadows			DE (CON)	DE
40A0	Subcontinental peri-Pannonic scrub			RO (PAN)	

A review of the evidence of the drivers of Annex I grassland habitat loss to arable conversion found the following:

- In Germany, there is evidence that the demand for crop feedstocks for bioenergy, principally the demand from biogas plants, has driven Annex I grassland losses from

Natura 2000 sites, particularly hay meadows, between 2004 and 2008-9 (Dieterich and Kannenwischer, 2012; NABU, 2014). In nine Natura 2000 sites in Baden-Württemberg surveyed in 2003-4 and again in 2008-9, 15% of the hay meadow area was no longer recorded (NABU, 2014), although there is only anecdotal evidence that this is due to bioenergy demands (Russi et al, 2016). A study found that the main driver for conversion of grassland to maize has been the need for silage maize for intensive dairy production and high grain maize prices for animal feed, but in some regions there was also documented conversion for bioenergy cropping (Laggner et al, 2014). The overall area of permanent grassland in Germany has stabilised since 2011 (BfN, 2017), but losses of Annex I grassland continue, as semi-natural grasslands continue to be lost because of grassland intensification (Benzler, Fuchs and Hünig, 2015). There are still incidents of illegal grassland conversion in Natura 2000 sites, for example in 2016 in the SAC 'Schönlager See, Jülchendorfer Holz und Wendorfer Buchen', which overlaps with the NSG 'Trockenhänge bei Jülchendorf und Schönlager See' in Mecklenburg-Vorpommern[58].

- Romania reported biofuel production as a pressure on subcontinental peri-Pannonic scrub in the pannonian biogeographic region, and also reported *Iris aphylla ssp. hungarica*, typical of subcontinental peri-Pannonic scrub, as under high pressure from arable conversion. However, this refers mainly to Natura 2000 sites where the habitat was reported upon declaration but is not actually present now; instead, the sites contain degraded pastures and woodlands, with observed losses of pasture areas to arable, including oilseed rape[59]. Romania has a support scheme for biofuel refineries[60], and has also had a major expansion of the area of oilseed rape cultivation in the last decade to 597,967 ha in 2017 (INSSE, 2017) [61], whilst the sunflower area has been maintained at around 100,000 ha since 2012. However, it is not clear whether the bioenergy demand is a key driver of this crop change. It is also not possible to ascertain whether this has directly led to the loss of peri-Pannonic scrub habitat and typical species.

- In Ireland, 28% of the surveyed area of hay meadow habitat was lost between 2015 and 2017 due to destruction for arable cultivation (Martin, O'Neill and Daly, 2018). As arable conversion is considered to be underreported due to survey limitations, it is

[58] NABU Mecklenburg-Vorpommern (25/4/2016) Illegal Grünland in Schutzgebiet umgebrochen. Available at https://mecklenburg-vorpommern.nabu.de/natur-und-landschaft/landnutzung/landwirtschaft/landwirtschaft-und-naturschutz/20605.html (accessed 3/3/2019)

[59] Personal communication, 25 January 2019, Mihai Pascu, executive director, Asociația pentru Promovarea Valorilor Naturale și Culturale ale Banatului și Crișanei ”Excelsior”

[60] Support scheme for processing of agricultural products in order to obtain biofuels, administered by the Directorate General for Rural Development – Managing Authority for the National Rural Development Programme (MA NRDP), as cited on p171 in Romania's National Renewable Energy Action Plan (NREAP) (2010), available at https://ec.europa.eu/energy/en/topics/renewable-energy/national-action-plans

[61] According to interviews with a number of large arable farmers, a part of the area reported as cultivated with oilseed rape is actually cultivated with a mixture of rape and another crop to fulfil the EFA requirement (cover crops option) in Romania (personal communication, Ralucca Barba, Highclere Consulting, 21 February 2019). As in practice the ratio of the two species in the actual seed mixture sown is at the discretion of the farmer (although both species should in theory be identifiable as present any on-spot field check carried out by the Paying Agency), it is likely that some EFA cover crops have been reported as oilseed rape.

likely that more may have been lost. However, none of the conversions were attributed to crop production for bioenergy, and there is currently very little arable cropping for bioenergy in Ireland[62]. This could change in the future as Ireland has ambitious bioenergy expansion plans (Department of Communications Climate Action & Environment, 2014; McEniry et al, 2013).

- Bulgaria reported arable conversion as causing the loss of four Annex I grassland habitat types but did not report biofuel production specifically as a pressure on these habitats. The sunflower production area in Bulgaria has increased rapidly, but most of this appears to be exported or used for food, and the Bulgarian biofuel industry is still very small, so it is not possible to draw conclusions as to the role of biofuel production as a driver[63].

- In Slovenia, large losses of Annex I grassland to arable have been reported in Natura 2000 sites (Zakšek et al, 2011). Slovenia reported a high pressure from loss of lowland hay meadows to arable conversion. No information was available on the drivers.

- In Slovakia, the Pannonic loess steppic grasslands are reported as being converted to arable. This is mainly due to historical drivers which have now lessened. However, there has been a continuing reduction in the registered permanent grassland area in Slovakia since 2003 (MARD SR, 2018). There have also been losses of meadows in SPAs due to arable expansions for the purpose of growing energy crops, for example in SPA Senianske rybníky (Gúgh et al, 2015). A subsidy programme incentivised the construction of 111 biogas plants between 2010 and 2015, and the biogas plants are primarily fuelled with maize (Kizeková et al, 2018). However, construction has markedly decreased since 2014 (due to a reduction in the subsidy) (Ladja, Lajdová and Bielik, 2015).

- In Belgium, the grasslands[64] under high pressure from arable conversion are small to very small areas surrounded by intensive farmland. These are mainly affected by conversion to crops directly or indirectly for animal feed, not bioenergy demands (Wibail et al, 2014).

In conclusion, grassland habitats obviously show maximum sensitivity to arable conversion as the habitat is essentially lost. Although there are only a few reported cases where the loss of HD habitats is specifically due to bioenergy cropping, this is likely to be because in many cases there are likely to be multiple drivers of arable conversion. This is supported by the evidence that arable conservation is a more common threat to HD habitats, and this is can be reliably assumed to be in part a direct and indirect result of the demand for bioenergy feedstocks.

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63 https://gain.fas.usda.gov/Recent%20GAIN%20Publications/Sunflower%20Market%20Diversification%20and%20Development_Sofia_Bulgaria_7-6-2015.pdf

64 hay meadows (6510) occupy less than 35 km2 in the Atlantic region and less than 119 km2 in the Continental region, Molinia grassland (6440) 0.6 km2 in the Atlantic region and 2.56 km2 in the Continental region, Nardus grassland (6230) 4 km2 in the Atlantic region (Belgium Article 17 report 2013)

Evidence of impacts of habitat conversion to arable on HD species

The loss of grassland habitat is already having significant negative impacts on many HD species. As for the intensification of grasslands discussed above, impacts are likely to be highest where conversion leads to the loss of old pasture where the grazing is extensive. The conversion of intensively managed grassland is expected to have less of an impact as it is not a suitable habitat for most farmland HD species. However, impacts may occur on all grasslands if conversion leads to the removal of hedgerows, ditches, and other linear features to create larger fields (i.e. consolidated fields). This removes important habitat for some species, such as for breeding and foraging (e.g. many bats).

The loss of heath or scrubland habitats to arable cropping is also recorded as a pressure on species in some Member States, but is rarely a medium or high pressure as heath and scrub are generally located on poor soils that are too wet, dry, steep, remote or rocky for crops.

Evidence of impacts of habitat conversion to arable on HD plants

Of the 281 HD vascular plants typical of grassland habitats, only Austria, Germany, Spain and Romania reported six species as under high pressure from arable conversion. Furthermore, the reliability and relevance of two of the cases is questionable as they have a highly restricted occurrence in habitats that are only marginally affected by arable farming[65]. The others are typical of dry or wet grassland, scrub or fallow habitats that are under pressure from intensification of arable or permanent cropping, although it is not possible to link this with bioenergy production[66]. The explanation for this lack of evidence of impacts of arable intensification is the fact that almost all of the HD vascular plant species have a restricted to extremely restricted occurrence, and most of the species occur in natural grasslands in mountains, coasts or other habitats relatively remote from arable agriculture.

Evidence of impacts of habitat conversion to arable on HD invertebrates

17 out of the 64 HD arthropod species associated with the cropland, grassland, heathland or sclerophyllous scrub ecosystems were reported by Member States as being under high

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[65] *Thesium ebracteatum* is found on only four sites on sandy, acidic, warm grasslands and heath within Natura 2000 sites in Germany, see https://ffh-anhang4.bfn.de/arten-anhang-iv-ffh-richtlinie/farn-und-bluetenpflanzen/vorblattloses-leinblatt-thesium-ebracteatum.html. *Sideritis javalambrensis* occurs only on the summits of the Sierra de Javalambre in Aragon, Spain, in degraded pine forests with *Juniperus sabina* or sparse dry grasslands on rocky soils, and is primarily under pressure from grazing activities, quarrying and the expansion of a skiing complex (IUCN red list assessment at https://www.iucnredlist.org/species/162220/5559977).

[66] *Boleum asperum* occurs only in two sites in Aragón and Cataluña in Spain, and is primarily under pressure from restructuring of agricultural land holdings, roads and motorways, but also from intensification of arable cropping on fallow land (IUCN red list assessment at https://www.iucnredlist.org/species/161856/5504642). The orchid *Himantoglossum adriaticum* is typical of vineyards and calcareous grasslands in Austria, and is mainly under pressure from abandonment. *Angelica palustris* in Romania is typical of wet meadows and fens, which are under pressure in the alpine and continental regions from conversion to arable (see for example http://www.ceeweb.org/wp-content/uploads/2015/09/Management-deficiencies-of-grassland-habitats-Mr-Adrian-Oprea.pdf). *Iris aphylla ssp. hungarica* is typical of subcontinental peri-Pannonic scrub as well as dry oak or beech forests and sunny meadows in Romania - see section on Annex I habitats for pressure on the habitat.

pressure from grassland conversion to arable (Figure 3-4). The Czech Republic reported ten HD arthropod species under pressure from grassland conversion to arable in the 2007 to 2012 period. This includes the Eurasian Toothed Grasshopper (*Stenobothrus eurasius*)[67] and the Predatory Bush Cricket (*Saga pedo*) (Holuša, Kočárek and Vlk, 2013), both typical of steppe grasslands. Similarly, *Carabus hungaricus* has a preference for overgrown steppe and fallow land, and is therefore particularly vulnerable to the expansion or re-expansion of arable cultivation (Cizek, Hauck and Pokluda, 2012). A number of other HD beetle and grasshopper species are grassland or shrub species that are found on remnants of grassland or open shrub vegetation in cropland mosaics, along field and road verges or hedges, and in small grassland patches. These species are vulnerable to frequent mowing for silage grass, afforestation, ploughing of grassland habitat, or destruction of field edges (Crișan et al, 2018; Krištín and Kaňuch, 2013; Marhoul and Olek, 2012; Rákosy, 2012; van Swaay et al, 2012).

Lepidopteran HD species associated with cropland, grassland, heathland or sclerophyllous scrub ecosystems are mainly affected by habitat loss and fragmentation, and secondarily by the lack of management of remaining habitat (Bubová et al, 2015). The Danube Clouded Yellow (*Colias myrmidone*) is reported as particularly under pressure from the transition from small grain mosaic management to more uniform and intensive agricultural systems, but also grassland loss to afforestation or conversion to arable (Rákosy, 2012). Landscape scale changes disrupt the proper functioning of the species' meta-populations, whilst habitat scale intensification of management eliminates larvae, suitable host plants and nectar sources for the adults (Marhoul and Olek, 2012). The butterflies *Euphydryas aurinia*, *Lycaena dispar*, *Maculinea arion*, *Maculinea nausithous*, *Maculinea teleius* are all associated with hay meadows and wet grasslands that are being lost to arable in the Czech Republic[68].

There is little information on how HD moths are potentially affected by bioenergy. Germany reported the moth *Proserpinus proserpina* as being under pressure from biofuel production in the Atlantic biogeographic region in the 2007 to 2012 period. The species can be affected by the loss of field margins, fallow and other habitat patches, such as meadow ditches and stream banks, resulting from arable intensification[69]. However, as the species has a very fluctuating population and occupation of sites, and is currently expanding in Germany, the pressure is low[70]. In Hungary and Romania, the moth *Cucullia mixta* and the beetle *Probaticus subrugosus* are associated with the Puzta grazing system and are reported as under pressure from arable conversion.

Evidence of impacts of habitat conversion to arable on HD freshwater species and amphibians

The negative effects of grassland conversion and maize cultivation on water quality of lowland rivers explains why Germany has reported biofuel production as a pressure on a freshwater fish (*Cottus gobio*) and mollusc (*Unio crassus*) typical of lowland rivers (habitat type 3260) and amphibian species associated with grassland and small waterbodies in agricultural areas

[67] LIFE project Lounske Stredohori Steppes at http://www.ochranaprirody.cz/en/life/life-lounske-stredohori-steppes/steppe-habitats-and-target-species/stenobothrus-eurasius/
[68] Article 17 report by Czech Republic for 2007 to 2012 period
[69] http://www.pyrgus.de/Proserpinus_proserpina_en.html
[70] https://ffh-anhang4.bfn.de/arten-anhang-iv-ffh-richtlinie/schmetterlinge/nachtkerzenschwaermer-proserpinus-proserpina.html

(*Rana temporaria, Rana dalmatina, Rana arvalis, Triturus cristatus, Pelobates fuscus, Bufo calamita*). Amphibians are strongly affected by declines in water quality (Egea-Serrano et al, 2012). Denmark reported biofuel production as a pressure on the freshwater Medicinal Leech (*Hirudo medicinalis*). The decline of leech populations in Europe is tied to the decline of their preferred amphibian host populations (Kutschera and Elliott, 2014), and Denmark reported an unfavourable or unknown status of 12 out of its 17 protected amphibians in the 2007 to 2012 period, with grassland conversion and water pollution as low level pressures[71].

Evidence of impacts of habitat conversion to arable on HD mammals

Member States reported 16 HD terrestrial **mammals** associated with cropland, grassland, scrub or heath (out of 63) as being under high pressure from conversion to arable, and 11 were reported as being under pressure from biofuel production (Figure 3-4).

The larger sized **bats** on the annexes associated with agricultural mosaics and grassland (e.g. horseshoe bats (*Rhinolophus* spp) forage primarily over pasture and open woodland patches, and are significantly adversely affected by loss of permanent pasture with wooded elements (hedges, tree lines, open and grazed woodland and wood pasture) (Dietz, Helversen and Nill, 2009). Other HD **mammals** associated with agricultural mosaics and cropland include the rodents European Hamster (*Cricetus cricetus*), Souslik (*Spermophilus citellus*) and Spotted Souslik (*Spermophilus suslicus*), and Severtov's Birch Mouse (*Sicista subtilis*), and the carnivorous mustelid European Polecat (*Mustela putorius*). These agricultural and cropland-associated species are characteristic of eastern European steppic grasslands, found in remnants of steppic vegetation on cropland and sometimes in perennial fodder crops (clovers, Lucerne etc). The most critical risk factor possibly associated with bioenergy cropping is the ploughing up of these vegetation remnants to expand arable cropping, and this was reported as a high-level pressure on these species in Romania, Bulgaria, Poland, Spain and Austria. Other critical risk factors are the loss of fallow fields and perennial fodder fields, or afforestation of grassland.

The species are under pressure as follows:

- European Black-bellied Hamster (*Cricetus cricetus*) is the only HD rodent that is truly typical of cropland as the species lives in burrows in deep soils typically used for arable cropping. However, hamsters rely on grassland, fallow or perennial fodder crops such as Lucerne, for feeding and refuge habitat after crop harvests in summer and autumn, which is critical for their reproductive success both to allow a second brood and to allow a diverse diet (La Haye et al, 2014; Tissier Mathilde et al, 2017). Romania reports the species as under pressure from biofuel production, and under a high pressure from grassland loss to arable. Germany and the Netherlands also report biofuel production as a pressure on the species. There is published evidence that maize monoculture is having significant adverse effects on the hamster populations in the Alsace region in France (Tissier et al, 2016) and the Hessen region in Germany (Albert, Reiners and Encarnação, 2011), and that at least some of this is associated with biogas feedstock production (Schumacher and Schultmann, 2017).

[71] ETC/BD (2015) Member States biogeographical assessments of conservation status of species and habitats under Article 17 of the Habitats Directive. Available at https://bd.eionet.europa.eu/article17/reports2012/

- The Romanian Hamster (*Mesocricetus newtoni*) is found in steppe grassland, Medicago, Taraxacum and cereal fields, vineyards, gardens, and scrubby slopes. Romania reports it as under medium pressure from biofuel production and under high pressure from grassland loss to arable and Bulgaria reports low pressure from biofuel production. Large monoculture fields of rapeseed have expanded in the hamster's distribution area in the Danube plain and Dobrudzha region, displacing the alfalfa and extensive cereal crops that are the species' suitable habitat (Nedyalkov et al, 2019)[72]. There are reports that pastures and long-term fallow fields have been ploughed in order to obtain CAP direct payments[73], which is likely to increase the pressure.

- Souslik (*Spermophilus citellus*) are under pressure from conversion of grasslands to arable land or their afforestation in various countries, and the Souslik EU species action plan published in 2013 identifies bioenergy cropping as a critical pressure in Austria and Bulgaria (Janák, Marhoul and Matějů, 2013). Bulgaria reported biofuel production as a high pressure in the Black Sea and Continental biogeographic regions, due to the expansion of oilseed rape and the loss of alfalfa, fallow and pasture[74]. However, the most critical pressures in its core habitat are related to lack of grassland management – such as low (insufficient) intensity of grazing or mowing, or a total absence of those – i.e. land abandonment. Small scale biogas production from grass cuttings could provide an incentive for the resumption of mowing.

- Spotted Souslik (*Spermophilus suslicus*) were reported by Poland as under pressure from agricultural intensification and conversion of grasslands into arable land in the 2007 to 2012 period. This mainly has an indirect effect by constraining the amount of potentially suitable habitat around the current populations, so they are unable to expand and suffer from overcrowding (Ziolek, Koziel and Czubla, 2017).

- The Steppe Polecat (*Mustela eversmanii*) is a typical species of Pannonic salt steppes and salt marshes (Šefferová, Janák and Ripka, 2008), but also occurs in pastures and cultivated fields in two population areas separated by the Carpathians. The main pressures on the species are the significant declines in the availability of its main prey Sousliks (*Spermophilus* spp.), hamsters and other small rodents (Šálek et al, 2013), associated with intensive agricultural production and the loss and degradation of steppe and grassland habitats, as described above.

- The Marbled Polecat (*Vormela peregusna*) is also reported as under high pressure due to loss of steppe grassland to arable cultivation in Bulgaria, presumably also due to the loss of its main prey Sousliks (*Spermophilus* spp.), hamsters and other small rodents, as described above.

[72] National statistics in Bulgaria show that in this region the oilseed rape area has increased around 20 times whilst the alfalfa area reduced by half (personal communication, Nedko Nedyalkov, researcher National Museum of Natural History – Bulgarian Academy of Science, 12 Jan 2019).
[73] personal communication, Nedko Nedyalkov, researcher National Museum of Natural History – Bulgarian Academy of Science, 12 Jan 2019
[74] personal communication, Nedko Nedyalkov, researcher National Museum of Natural History – Bulgarian Academy of Science, 12 Jan 2019

- The two birch mouse HD species, the Northern Birch Mouse (*Sicista betulina*) and Severtzov's Birch Mouse (*Sicista subtilis*), are among the rarest and least studied small mammals in Europe and have fragmented and poorly known distributions and habitat preferences. The Northern Birch Mouse (*Sicista betulina*) in Schleswig-Holstein, northern Germany, might be under pressure from arable conversion, as reported by Germany, but no information is available (Meinig, Schulz and Kraft, 2015). Severtzov's Birch Mouse (*Sicista subtilis*) in Romania has a confirmed population in one Natura 2000 site, where its habitat is reported as being ploughed up by local farmers or overgrazed by sheep (Cserkész et al, 2015). The only Hungarian population in contrast has a quite high density in a habitat complex with steppe grasslands and abandoned arable fields (Cserkesz, 2007). Given the limited information, it is not possible to draw wider conclusions about these two species' vulnerability to bioenergy production.

Figure 3-4 HD species reported as under pressure from biofuel production and/or high pressure from grassland conversion to arable; and/or under pressure according to scientific literature

Key to biogeographical regions: ALP Alpine; ATL Atlantic; BLS Black Sea; BOR Boreal; CON Continental; MAC Macaronesian; MED Mediterranean; PAN Pannonian; STE Steppic.
MS = Member State H = high M = medium L = low
Evidence of sensitivity according to the literature – see text for related reference sources.

Taxonomic group	**Species name**	**MS reporting high pressure from grassland conversion to arable**	**MS reporting biofuel production pressure**			**Evidence in the literature re grassland conversion to arable**
			H	**M**	**L**	
Plant	*Angelica palustris*	RO (ALP, CON)				
Plant	*Boleum asperum*	ES (MED)				
Plant	*Himantoglossum adriaticum*	AT (CON)				
Plant	*Iris aphylla ssp. hungarica*	RO (CON)				
Plant	*Sideritis javalambrensis*	ES (MED)				
Plant	*Thesium ebracteatum*	DE (CON)				
Arthropod (beetle)	*Pilemia tigrina*	HU (PAN)				RO
Arthropod (beetle)	*Probaticus subrugosus*	HU (PAN)				
Arthropod (beetle)	*Carabus hungaricus*	CZ (PAN)				CZ
Arthropod (beetle)	*Carabus menetriesi pacholei*	CZ (CON)				
Arthropod (beetle)	*Bolbelasmus unicornis*	CZ (PAN)				
Arthropod	*Stenobothrus eurasius*	CZ (CON)				CZ
Arthropod	*Saga pedo*	CZ (PAN), IT (CON)				CZ
Arthropod	*Apteromantis aptera*	ES (MED)				
Arthropod (butterfly)	*Coenonympha hero*	LT (BOR)				

Taxonomic group	Species name	MS reporting high pressure from grassland conversion to arable	MS reporting biofuel production pressure			Evidence in the literature re grassland conversion to arable
			H	M	L	
Arthropod (butterfly)	*Colias myrmidone*					Yes
Arthropod (butterfly)	*Euphydryas aurinia*	CZ (CON)				
Arthropod (butterfly)	*Lycaena dispar*	CZ (CON,PAN)				
Arthropod (butterfly)	*Lycaena helle*	LU (CON)				
Arthropod (butterfly)	*Maculinea arion*	CZ (CON)				
Arthropod (butterfly)	*Maculinea nausithous*	CZ (CON,PAN)				
Arthropod (butterfly)	*Maculinea teleius*	CZ (CON,PAN)				
Arthropod (moth)	*Cucullia mixta*	RO (CON)				
Arthropod (moth)	*Proserpinus proserpina*		DE(ATL)			
Mollusc	*Vertigo geyeri*	IT (ALP)				
Mollusc	*Unio crassus*			DE(ATL)		
Other invertebrate	*Hirudo medicinalis*	DK (CON)				
Amphibian	*Triturus cristatus*	DE (CON)			DE(ATL)	
Amphibian	*Hyla arborea*			DE(ATL)		
Amphibian	*Pelobates fuscus*		DE(ATL)			
Amphibian	*Bufo calamita*			DE(ATL)		
Amphibian	*Rana arvalis*				DE(ATL)	
Amphibian	*Rana dalmatina*				DE(ATL)	
Amphibian	*Rana temporaria*			DE(ATL)		
Fish	*Cottus gobio*				DE(ATL)	
Mammal	*Cricetus cricetus*	RO (ALP, CON, STE)		DE(ATL), NL(ATL)	DE(CON), RO(PAN, STE)	FR, DE
Mammal	*Mesocricetus newtoni*	RO (STE)		RO(STE)	BG (CON, BLS)	BG
Mammal	*Microtus cabrerae*	ES (MED)				
Mammal	*Mustela eversmanii*	BG (BLS,CON)				Yes
Mammal	*Sicista betulina*				DE(CON)	
Mammal	*Spermophilus citellus*	AT(CON), RO (ALP,BLS,CON,PAN,STE)	BG(BLS, CON)			AT, BG
Mammal	*Spermophilus suslicus*	PL (CON)				PL
Mammal	*Vormela peregusna*	BG (ALP,BLS,CON)				BG
Mammal	*Eptesicus serotinus*			DE(ATL)	DE(CON)	
Mammal (Bat)	*Myotis brandtii*				DE(ATL)	
Mammal (Bat)	*Myotis myotis*			DE(ATL)		
Mammal (Bat)	*Myotis mystacinus*			DE(ATL)		
Mammal (Bat)	*Myotis nattereri*			DE(ATL)		
Mammal (Bat)l	*Plecotus auritus*			DE(ATL)		
Mammal (Bat)	*Plecotus austriacus*			DE(ATL)		
Mammal (Bat)	*Rhinolophus ferrumequinum*			DE(CON)		

Evidence of impacts of habitat conversion to arable on BD birds

The loss of grasslands, and especially semi-natural grasslands, to bioenergy crops can be reliably expected to have a substantial negative impact on a wide range of BD birds, as their structure and diversity, vegetation and associated invertebrates and other animals provide the special habitat and food requirements required by many scarce and rare species. This has been shown by an analysis of BD birds that reveals that some 54 species regularly occur in natural and semi-natural habitats influenced by agriculture, as indicated in Figure 3-5 and in Annex 1 of this report. The table summarises the number of BD birds that have more than 10% of their European population in one or more agricultural habitat types according to Birdlife International (Tucker and Evans, 1997). In total, 62 out of the 195 BD birds are considered to be agricultural species. The findings indicate that semi-natural grassland habitats are particularly important, especially steppe grasslands as they are relatively restricted in the EU yet still hold significant populations of 32 key agricultural bird species. Although arable and improved grasslands also hold 32 species, this is probably largely due to the large area of the habitat (due to the well-known species-area relationship). In addition, many of these species are dependent on extensive dry cereal production. Semi-natural Mediterranean shrublands and wet grasslands are also particularly important habitats, considering their relatively small extent. Whilst this analysis is now rather old, there is no reason to believe that the situation has changed substantially.

Figure 3-5 Totals of agricultural species listed on Annex I of the Birds Directive according to habitat type and conservation status category

Key: Moor = grazed moorland and tundra; Med = grazed Mediterranean shrublands; AIG = arable and improved grasslands; SG = steppe grasslands; MG = montane grasslands; WG = wet grasslands; PC = permanent crops; PW = pastoral woodlands. NB. The number of species occurring in all habitats is not equal to the sum of totals in each habitat, because many species occur in more than one habitat type. Habitat use relates to the proportion of the European population that is estimated to occur within each habitat.

	All	Moor	Med	AIG	SG	MG	WG	PC	PW
Total agricultural species	62	3	21	32	32	6	13	5	12
Habitat use									
• 10-75% of population		0	11	22	21	5	10	5	11
• >75% of population		3	10	10	11	1	3	0	1

Source: adapted from (Tucker and Evans, 1997)

Member States have reported high pressure on BD birds from the conversion of grasslands and fallow to crops in 2007-2012, as indicated in Figure 3-6, and there is also a documented case in the literature from Germany. In the Hellwegborde SPA in northern Germany, conversion of arable fallow to maize cultivation for biogas is reported as one of the main causes of continued population declines of Corncrake (*Crex crex*), and Montagu's Harrier (*Circus pygargus*) (Joest, 2013). A modelling study in Germany also predicted that further increase in maize production would have a significant negative impact on the populations of Red-backed Shrike (*Lanius collurio*) and Woodlark (*Lullula arborea*) (Sauerbrei et al, 2014).

The literature indicates a high sensitivity of BD birds to grassland and fallow conversion to arable, and to intensification of arable cropping. In southern Portugal, the introduction of irrigation has reduced overall landscape suitability for Little Bustard *Tetrax tetrax tetrax* by replacing grasslands (pastures and fallows) and low intensity permanent crops with irrigated annual crops (Moreira et al, 2012) (although it is not known whether this crop change is related to demands for bioenergy crops or not). Similarly, ploughing of pasture and subsequent irrigation have reduced available habitat of the Black-bellied Sandgrouse (*Pterocles orientalis*) on the Iberian Peninsula, while the removal of marginal areas of semi-natural vegetation and the increased application of agro-chemicals have reduced food availability, and it is likely these practices have been responsible for some local extinctions. The Calandra Lark (*Melanocorypha calandra)* has suffered declines in Southern Europe due to the loss of fallows and land use changes away from cereal crops. Similarly, the main threats to the Greater Short-toed Lark *(Calandrella brachydactyla)* are from agricultural intensification leading to loss of fallows and increased crops area. The vulnerable Great Bustard (*Otis tarda*) is threatened by increased habitat degradation, fragmentation and loss due to agricultural intensification and land-use changes which continues as a result of ploughing of grasslands and intensive grazing (Birdlife International, 2015).

Fallow land with self-regenerated vegetation has been shown to be a particularly important habitat for birds, both for ground nesting species and as feeding habitat (Dicks et al, 2013; Herzon et al, 2011). This is particularly so for birds of extensively managed dry cereals as the incorporation of significant areas of fallow cropland (which is often grazed) is an important part of the farming system. Although now scarce in Europe, particularly important areas remain in dry areas of Spain and are of very high conservation importance, as they hold large proportions of some BD birds, including Great Bustard (*Otis tarda*), Lesser Kestrel (*Falco naumanni*) and other European threatened species (Bota et al, 2005; Delgado and Moreira, 2000; Suárez, Naveso and de Juana, 1997; Tucker and Evans, 1997). Clearly the use of fallow land for crops for bioenergy will be detrimental for a wide range of BD birds, but especially those of the remaining areas of dry cereal. This has been found for the Lesser Kestrel (*Falco naumanni*), which has a lower breeding success in regions where fallow within cereal rotations has been lost (Catry et al, 2012; Catry et al, 2013). Displaying Little Bustard (*Tetrax tetrax tetrax*) males are more abundant in landscapes with high proportions of fallow with legume and weedy vegetation than in intensive arable areas (Faria, Rabaça and Morales, 2012; Santangeli and Dolman, 2011). Great Bustard (*Otis tarda*) also show a clear preference for nesting in cereal fields and young fallows compared to other habitats (Rocha, Morales and Moreira, 2012).

A number of Bunting species (*Emberizidae*) are suspected to be in decline owing to a reduction in cultivation of cereal crops and the intensification of farmland management, including that for bioenergy. The BD Annex I listed Ortolan Bunting (*Emberiza hortulana)* is under pressure due to ongoing conversion of relatively extensively used habitat to crop fields for biofuel (Birdlife International, 2015). In eastern Germany, the intensification of agriculture, ploughing of grassland and disappearance of large areas of low-intensity pastures and unproductive grassland have driven habitat loss for the Barred Warbler *(Sylvia nisoria)*[75].

In contrast, some birds associated with open land can show positive redistribution responses to moderate decreases in the proportion of grassland and corresponding increases in arable

[75] BirdLife International (2019) BirdLife Data Zone *Sylvia nisoria*. http://datazone.birdlife.org/species/factsheet/barred-warbler-sylvia-nisoria

land at a landscape scale (Sauerbrei et al, 2014). This is because the increase in crop diversity, and therefore increase in structural diversity (e.g. in crop height and density) and associated food diversity (e.g. plant seeds, invertebrates) in turn leads to an increase in breeding and feeding opportunities for some birds. This benefits species that feed on a mixture of seeds and insects over the year, such as the Ortolan Bunting (*Emberiza hortulana*), which favours areas of bare soil and legume forage crops (Fonderflick et al, 2010; Morelli, 2012). However, most bird species that can gain from such changes are likely to be generalist widespread species, and not BD species. Furthermore, positive impacts are dependent on moderate changes and the replacement of improved (rather than semi-natural) grassland with a mixture of relatively extensively managed crops (e.g. including legumes and oil-seed rape) and not maize, which supports a highly impoverished fauna, including birds (Klenke, Frey and Zarzycka, 2017).

Many non-Annex I birds are also affected, and consequently, in Europe, bioenergy is predicted to have a negative impact on a larger proportion of species (96% of all bird species) compared to climate change alone (average of 37% across scenarios) which poses a direct threat to farmland birds (Meller et al, 2015).

Figure 3-6 BD birds reported as under pressure from biofuel production and/or high pressure from grassland conversion to arable; and/or under pressure according to scientific literature

MS = Member State H = high M = medium L = low
Evidence of sensitivity according to the literature – see text for related reference sources.

Species name	MS reporting high pressure from grassland conversion to arable	MS reporting biofuel production pressure			Evidence in the literature for either pressure
		H	M	L	
Accipiter brevipes				RO	
Acrocephalus paludicola	LT				
Alcedo atthis				LU	
Anser erythropus			BG		
Anthus campestris	AT				
Aquila chrysaetos	BG				
Aquila heliaca	BG, RO				
Aquila pomarina	BG, LV, RO, SI				
Asio flammeus	RO				
Branta ruficollis			BG		
Burhinus oedicnemus	AT, RO				
Buteo rufinus	BG			RO	
Calandrella brachydactyla	BG, RO				
Calidris alpina schinzii	LT				
Chersophilus duponti	ES				
Ciconia ciconia ciconia	AT, HU, IT, LU, SI				
Ciconia nigra			RO		
Circaetus gallicus	RO				
Circus aeruginosus	LU		LU		
Circus cyaneus	LU		LU		
Circus pygargus	IT, LT, RO		BG		DE
Coracias garrulus	LT, SI				
Crex crex	AT, BE, BG, LT, LU, LV, RO, SI		LU		DE
Dendrocopos leucotos		BG			
Emberiza hortulana	BG				

Species name	MS reporting high pressure from grassland conversion to arable	MS reporting biofuel production pressure			Evidence in the literature for either pressure
		H	M	L	
Falco cherrug	RO				
Falco naumanni	BG		BG		
Falco vespertinus	BG, IT, RO			BG	
Gallinago media	FI, LT				
Hieraaetus pennatus	BG				
Lanius collurio	BE, LT, LU, SI			LU	DE
Lullula arborea	BG				DE
Luscinia svecica cyanecula				LU	
Melanocorypha calandra	BG, IT				
Milvus migrans	BG			LU	
Milvus milvus	BG, LU		LU		
Pernis apivorus	BG, LT, SI				
Philomachus pugnax	LT				
Sylvia nisoria	AT, BG				
Tetrao tetrix tetrix	LT				

3.2.3 Intensification of grassland management

General biodiversity impacts

The intensification of grassland management (e.g. drainage, re-seeding, higher rates of fertiliser use and herbicide use) may be driven by bioenergy demands where grass silage is used as a feedstock for biogas. However, currently silage is only being used for this purpose in small volumes in some regions, e.g. in Germany (Daniel-Gromke et al, 2018) and Slovakia (Kizeková et al, 2018). In Germany, the grass silage used for bioenergy most often composes just the third and fourth grass cuttings as the previous cuttings are used for animal feed (Daniel-Gromke et al, 2018). Therefore, bioenergy is currently not an important direct driver of intensification of grassland management, although the potential for grass silage to be used as a bioenergy feedstock in the future has been highlighted by various studies (Cosentino et al, 2014; McEniry et al, 2013).

As discussed in Section 2, however, bioenergy markets are likely to be an indirect driver of intensification of grassland management to some extent, along with the pressure to convert grassland into arable land for the production of animal feed (Laggner et al, 2014). There is therefore an indirect land use change impact, although this is very difficult to quantify.

As mentioned in the section above, the biodiversity importance of an agricultural area (e.g. in terms of its diversity and abundance of characteristic species) and, in particular, threatened species that are the focus of EU conservation objectives) of farmland declines with increasing agricultural improvement and intensification (Aebischer, 1991; Billeter et al, 2008; Donald, 1998; Donald, Green and Heath, 2001; Poláková et al, 2011). Based on a review of evidence, Polakova et al (2011) concluded that the following agricultural management practices on grassland have the most important ecological impacts on species:

- Use of fertilisers, which results in fast growing and dense grasslands that provide poor breeding and feeding habitats, as well as reducing plant species diversity and associated animal communities.

- Ploughing and re-seeding of grasslands with grass cultivars, which further reduces plant species diversity (and associated animal communities), increases the density and growth rates of the grassland, and impacts soil biodiversity.

- High grazing densities, which reduces plant species and structural diversity in pastures, and the abundance and diversity of associated animal communities; as well causing high losses of ground nesting birds.

- Cutting for silage, which results in the loss of species-rich hay meadows (where practised) and high losses of ground nesting birds and other animals.

- Use of herbicides (and other forms of weed control), which further reduces plant species diversity in grasslands (and associated animal communities).

- Removal of boundary habitats (such as hedgerows, stone terraces and ditches) and other non-farmed habitats (woodlots, trees and ponds), which reduces habitat diversity and connectivity amongst non-farmed habitat patches in the farmed landscape.

Other agricultural related contributory causes include high predation rates as a result of high levels of disturbance (which expose nests to predation and interfere with predator defence behaviours).

Furthermore, although other non-agricultural factors affect farmland habitats and their species (e.g. high predator densities, alien invasive species, hunting, collisions with power lines etc, external pollution sources), the available evidence indicates that agricultural factors have the most significant impact on population trends in most farmland species.

It should be noted that an important driver of the intensification of semi-natural grasslands is their limited profitability and therefore economic unviability. As a result farmers may find themselves in a situation where they have to either improve their land and intensify their farming or abandon it, which is also detrimental for most HD habitats and their associated species of high conservation importance (Macdonald et al, 2000; Poláková et al, 2011; Stoate et al, 2009). Using extensively harvested grass cuttings for bioenergy could help to increase the economic sustainability of the management of some HD grasslands, where suited to their conservation management requirements, thereby benefiting biodiversity by preventing abandonment (Heinsoo et al, 2010). However, such a positive biodiversity impact is only achieved if the management remains extensive and no extra fertilisation or other inputs are added, and also only if the timing and system of cutting does not have a negative impact on bird breeding or invertebrate populations.

Evidence of impacts of grassland intensification on HD habitats

As all HD agricultural habitats are natural or semi-natural habitats they will be affected by the agricultural improvement and intensification of grassland use, with more frequent cutting, machinery passes and fertiliser applications etc. This is indicated in the EEA's analysis of Member State reporting as set out in the State of Nature Report (EEA, 2015b). For example, lowland hay meadows can in some cases tolerate a low level of fertilisation from animal manure and two cuts, but only where eutrophication through nitrogen deposition or previous fertilisation is not already affecting the grasslands, and in northern Europe all fertilisation is considered as negative for habitat quality (European Commission, 2014). Sedge meadows and *Molinia* meadows can tolerate two cuts, and sometimes two cuts are needed for a period to restore the species richness of abandoned grasslands of these types (Cop, Vidrih and Hacin, 2009; Šefferová Stanová and Čierna, 2011). Other grassland types will only tolerate one late cut.

Evidence of impacts of grassland intensification on HD species and BD birds

Intensification of grassland management with more frequent cutting, machinery passes, and fertiliser application makes the habitat unsuitable for most HD and BD grassland species. Member States reported that grassland-related birds are the most vulnerable to the modification of cultivation practices, including, for instance, agricultural intensification, grassland removal for arable land, and crop change, and for the HD species this is the second most important pressure after abandonment, as set out in the State of Nature Report (EEA, 2015b).

Arthropods are particularly affected by grassland intensification, especially butterflies which lose their host plants that cannot survive the frequent cutting, and also suffer direct mortality of their larvae and eggs and lack of nectar resources for the adults (Cizek et al, 2012). In particular, large scale and comprehensive mowing of grassland can extinguish grassland butterfly populations, even if the other intensification factors are not present (e.g. fertiliser and herbicide use are avoided), as the example of *Colias myrmidone* in the Czech Republic shows (Konvicka et al, 2008). Intensification of grassland management also makes the habitat unsuitable for amphibians and reptiles.

The larger bat species are all substantially affected by declines in the abundance of their main prey (large beetles and moths) from the loss of animal dung on grazed pastures or by the use of avermectins (i.e. drugs and pesticides that are used to treat grazing livestock against parasites), which also has an impact on bats' prey (Dietz, Helversen and Nill, 2009). They are therefore adversely affected by grassland intensification, which results in more grassland being used for silage production and a reduction in livestock grazing on pasture. For example, the Greater Horseshoe Bat (*Rhinolophus ferrimequinum*) population has suffered severe declines in Europe in the last decades partly due to changes in livestock grazing practices[76].

BD birds that are particularly negatively affected by grassland management intensification include the Corncrake (*Crex crex*), which is particularly affected by switches from hay to silage which produces vegetation that is not very suitable for the species. Furthermore, silage is cut earlier and by faster machines, which results in high rates of mortality of young and adults. This is one of the key factors that has given rise to the widespread decline of the species across much of Europe where intensification has been most prevalent (Crockford et al, 1997).

[76] Hutson, T, Spitzenberger, F, Juste, J, Aulagnier, S, Fernandes, M, Alcaldé, J T (2007) *Rhinolophus ferrumequinum*. The IUCN Red List of Threatened Species 2007: e.T19517A8948327. Downloaded on 24 August 2018.

Similarly, the cutting of grass and cereal silage occurs during the breeding season of the ground nesting Ortolan Bunting (*Emberiza hortulana*), which has led to particularly negative effects on this species (Dieterich et al, 2016).

3.2.4 Crop change on existing arable land and associated arable intensification

General biodiversity impacts

Crop changes on existing arable land can result in changes in fertiliser use, pesticide use, tillage, cropping season, changes in crop types and rotations. Bioenergy demand has partly driven increases in cultivation of maize and rapeseed on existing arable land by replacing other crops. For example, in Romania, the increased surface area of rapeseed and sunflower has replaced cereals (Vasile et al, 2016). The increase in certain crops for bioenergy can also lead to an indirect intensification of arable production in other areas. As mentioned in the section above, the biodiversity importance of an agricultural area (e.g. in terms of its diversity and abundance of characteristic species) and, in particular, threatened species that are the focus of EU conservation objectives) of farmland declines with increasing agricultural improvement and intensification (Aebischer, 1991; Billeter et al, 2008; Donald, 1998; Donald, Green and Heath, 2001; Poláková et al, 2011). The agricultural management practices associated with arable crop changes and arable intensification that have the most important ecological impacts on species, according to a review of evidence in Polakova et al (2011) are:

- Use of fertilisers, which results in fast growing and dense crops that provide poor breeding and feeding habitats, as well as reducing plant species diversity and associated animal communities.

- Use of herbicides (and other forms of weed control), which further reduces plant species diversity in crops (and associated animal communities).

- Use of pesticides, which reduces the abundance and diversity of invertebrate food resources for invertebrate predators (eg other invertebrates, birds, bats etc).

- Crop specialisation and reduced crop rotations, which reduces structural and ecological heterogeneity in the landscape resulting in reduced breeding and feeding options and reduced ecological connectivity amongst habitat patches.

- Changes in timing of agricultural practices, such as crop sowing, which, for example, affects the availability of over winter food and suitable vegetation (in terms of height and density) for breeding.

Other agricultural related contributory causes include increased mechanisation/efficiency of cropping (e.g. leading to less spilt grain) and high predation rates as a result of high levels of disturbance (which expose nests to predation and interfere with predator defence behaviours).

The change of **cereals to oilseed rape** or sunflower generally means a substantial increase in pesticide use. Rape has the highest insecticide application rate of all arable crops because of its attractiveness for insect pests if grown continually on the same land, and also requires fungicide applications (Garthwaite et al, 2014; Rossberg, 2016). The higher insecticide use on oil crops compared to cereal crops is also likely to negatively impact on invertebrates in the field margins, such as grasshoppers (Bundschuh et al, 2012), which will have a knock-on effect on insectivorous vertebrates of conservation concern. Pesticide applications on oilseed rape destined for biodiesel do not differ from other oilseed rape fields as a certain yield quality must still be met (DBFZ, 2014). Oilseed rape can provide more food resources to foraging insects, birds and mammals than cereal crops, in the form of pollen, invertebrates, and seeds, but the resources are quickly removed at harvest in June, so do not necessarily result in a population increase unless followed by other flowering crops or compensated by sufficient semi-natural vegetation (Riedinger et al, 2014). It is therefore difficult to assess the overall effects of increases in oil-seed rape on biodiversity, but they are likely to be relatively low as intensive agriculture is already of very low value for EU protected habitats and species. But there may be benefits for some species when a switch to oilseed rape significantly increases the crop diversity within a landscape.

The change of **cereals to maize** is associated with more soil erosion from rain and runoff due to the greater exposure of soil in the rows between plants and the lack of plant cover, except in the rare cases where intercrops or undersown cover crops are used. Maize can be grown in monoculture year after year, unlike cereals and oil crops. Earthworm abundance tends to be lower in maize than in other crops because of the more intensive tillage (Felten and Emmerling, 2011). Maize crops on average receive a higher input of fertiliser and herbicides than cereal crops, and maize is liable to decrease soil organic matter as the crop is more demanding than cereals and leaves little residue in the soil after harvest. As there are no significant differences between the management of maize for animal feed and maize for biogas (DBFZ, 2014), the biodiversity impact of bioenergy maize is similar to that of any maize crop. Farmers generally use insecticide treated maize seed during the conversion of an arable crop rotation to maize monoculture to avoid soil pests (Kathage et al, 2018). For these reasons, maize supports an invertebrate community with very low diversity and biomass, or with occasional short-lived flushes of higher biomass (Klenke, Frey and Zarzycka, 2017). For example, a German study has shown that bumblebees have a lower population growth in maize-dominated landscapes, resulting from the low pollen diversity available in such landscapes (Hass et al, 2018).

The change of **cereals to sugarbeet** is associated with high levels of herbicide use, but also a long cropping period with a high proportion of open ground and a more diverse weed flora. Sugarbeet therefore offers relatively good breeding habitat for ground-nesting birds compared to cereals, and also winter feeding habitat (Gillings et al, 2009).

Maize and sugarbeet sown as spring crops need much more irrigation water than the winter crops (wheat, rape), because of the summer growing season (ETC/SIA, 2013). Irrigation leads to rapidly growing and dense crops, requiring high levels of fertiliser use and pesticides, which further reduces their suitability for most wild arable pants and associated invertebrates and other animals. Water abstraction for irrigation can also lead to impacts on wetland habitats, which can have profound impacts on associated species (Amores et al, 2013).

Cereals grown for bioenergy will receive fewer pesticide applications than cereals grown for food; however, in most cases farmers do not decide on the market for their cereal crop until shortly before harvest when they know the quality, and therefore do not reduce pesticide use earlier in the season (DBFZ, 2014).

Evidence of impacts of crop change and arable intensification on HD plants

The four HD plants associated with cropland occur as weeds in extensively managed cereal (wheat and oat) fields and are eradicated if arable cultivation is intensified with fertilisers and dense crop growth, herbicide use, and the sowing of temporary grasslands in rotations. All these species are expected to be negatively affected if the traditional low intensity cereal production is replaced by higher intensity bioenergy food crops such as maize.

In Wallonia, Belgium, the extensive cereal arable habitat of the plant *Bromus grossus* is being replaced by maize cultivation and temporary grassland (Wibail et al, 2014). In Hessen in Germany, *Notothylas orbicularis* is under pressure from intensification of cereal cropping with a second autumn crop, partly driven by biogas demands (Drehwald, 2012). *Agrimonia pilosa* is a grassland species which occurs on cropland edges in eastern Europe[77], and is therefore adversely affected by all arable intensification measures that remove or adversely affect edge features (for example by field expansion or increased ploughing or herbicide treatment of edges, or increased fertiliser or pesticide use that spills over onto grassed edges. *Linaria ricardoi* is associated with weed communities in autumn and winter cereal crops in Portugal, but has declined dramatically due to the use of herbicides in cereals; now most populations occur in ruderal areas, such as on slopes and roadsides and in olive groves, where herbicide use is infrequent (ICN, 2007). It is similarly at risk from all crop intensification measures that remove or adversely affect such refuge habitats.

Evidence of impacts of crop change and arable intensification on HD invertebrates

Hungary reported biofuel production as a high-level pressure on the beetle *Pilemia tigrina*, which is a species considered typical of cropland mosaics (as shown in Figure 3-4). *Pilemia tigrina* is found along field margins with remnants of steppe grassland vegetation containing its host plant *Anchusa barrelieri*, and it is under pressure in Romania from the combined effects of loss of field margins to arable intensification and scrub and invasive species encroachment due to abandonment of management (Crișan et al, 2018). Presumably, the pressures on the beetle in Hungary are similar (Tóth et al, 2016).

The other HD invertebrates associated with cropland mosaics (*Colias myrmidone, Erebia christi, Eriogaster catax, Paracaloptenus caloptenoides*) are affected by arable intensification through the associated loss of remnants of grassland or open shrub vegetation in cropland mosaics, along field and road verges or hedges, and in small grassland patches.

Evidence of impacts of crop change and arable intensification on HD reptiles and amphibians

[77] http://eseis.ut.ee/efloora/Eesti-vte/species/Agrimonia_pilosa.html

Two HD reptile species are particularly associated with the cropland ecosystem. They are unlikely to be directly affected by bioenergy cropping but could be affected by agricultural intensification. The Maltese Wall Lizard population occurs on traditionally cultivated land and gardens, but most of the population occurs on scrubland and rocky areas on islands where it is unlikely to be affected by bioenergy cropping, and the main pressures on the species are not related to agriculture (Rodríguez et al, 2014). The Milos Wall Lizard population on the Milos islands of Greece is found mostly in sparsely vegetated habitats not associated with cropland, but does also occur in vineyards and barley fields where intensification of production or conversion to plantations or short rotation coppice would negatively affect the species.

Germany reported that the Common Spadefoot Toad (*Pelobates fuscus*) is under pressure from the impacts of increased biofuel production in the 2007-2012 period (BfN and BMUB, 2014). This species and another HD amphibian, Green Toad (*Bufotes viridis*), are associated with cropland because they occur in a wide range of modified habitats, including gardens and mixed agricultural areas - and often benefit from disturbed habitats. They are indirectly affected by agricultural intensification through the impact of pesticide use on their prey, intensive grassland use (overstocking and frequent machine treatments), and impacts on water quality from soil erosion or pesticide run off (Wagner et al, 2014).

Evidence of impacts of crop change and arable intensification on HD mammals

Ten **bat** species are associated with the cropland ecosystem because they forage across arable fields or along tree lines and hedges in mosaic agricultural landscapes (Dietz, Helversen and Nill, 2009). The larger bat species are all substantially affected by declines in abundance of their main prey (large beetles and moths) from pesticide use on cereal crops and the loss of animal dung or use of avermectins to treat livestock. Changes in arable crop are not likely to cause significant changes in bat feeding behaviour, but the removal of linear features to make fields larger has a significant impact on bats. They could benefit slightly from increased abundance of flower-visiting insects on oilseed rape or be more adversely affected by increased maize if it results in decreased soil invertebrate populations.

European Hamsters (*Cricetus cricetus*) are the only HD rodent that are truly typical of cropland as they live in burrows in deep soils typically used for arable cropping. In Western Europe, they are severely under pressure from early crop harvesting, the loss of long-term fallows and perennial legume crops from arable rotations, and the conversion of cereal rotations to monoculture maize (La Haye et al, 2014; O'Brien, 2015; Tissier et al, 2016). Hamsters prefer crops that provide springtime cover and forage opportunities, especially winter cereal crops and lucerne (*Medicago sativa*) (Orbicon Écosphère ATECMA Ecosystems LTD, 2009). In contrast, maize and root crops provide no cover in spring, leaving them highly vulnerable to predation. The simplification of crop rotations and monoculture means that large areas are harvested or ploughed at the same time, leaving no suitable refuges. Ploughing of stubble directly after harvest removes autumn food sources, and deep ploughing destroys hamster burrows. Romania, Germany, and the Netherlands all report the species as under pressure from biofuel production (see previous section for discussion).

Evidence of impacts of crop change and arable intensification on BD birds

Bird species select crops as breeding or foraging habitat in relation to the vegetation structure of the crops and their dynamics, the composition and abundance of weed vegetation within a crop, and the influence of vegetation height and density on predation risk and food search times (Glemnitz, Zander and Stachow, 2015). Consequently, the type and timing of agricultural practices (e.g. using or avoiding ploughing) and the timing of sowing, can decide whether the crops provide suitable breeding habitat. Similarly, food availability is significantly affected by changes between crop species and cropping seasons, and also by herbicide and insecticide use in crops (Chamberlain et al, 1999; Ewald et al, 2016; Geiger et al, 2010).

As a result of the above factors, maize crops provide unsuitable breeding habitat for most species of birds. Although the high proportion of bare soil in young maize fields attracts ground nesting birds (e.g. Lapwing, *Vanellus vanellus*), this becomes unsuitable by June due to the rapid and dense crop growth. Bird breeding success in maize is also negatively affected by the lack of weeds and frequent herbicide applications (Tillmann, 2011). Due to the lack of weeds and the impoverished invertebrate fauna (see above), maize fields also provide poor foraging areas for birds. Winter maize stubbles can attract foraging birds, but they contain fewer food resources than cereal stubbles, and most maize stubbles are ploughed in straight after harvest anyway (Dieterich et al, 2016). A modelling study in Germany predicted that further increase in maize production, from 2.6 to 2.9, 3.6 and 4.3 million ha, would have a significant negative impact on the populations of the BD Annex I species Red-backed Shrike (*Lanius collurio*) and Woodlark (*Lullula arborea*) (Sauerbrei et al, 2014). Another model concluded that replacing maize for energy production on 15% and 30% of the area covered by other cash crops had a negative effect on the population sizes of Skylark (*Alauda arvensis*), and an aggregated cultivation of maize amplified these effects on the species (Gevers et al, 2011).

Thus, in general, the conversion of cereals, oil-seed rape and other arable crops to maize is detrimental for the farmland bird community. However, it is important to note that conventionally managed arable crops are also poor habitat for most farmland species, and within the main areas that are subject to maize growing for bioenergy, generally do not hold significant populations of any BD birds, with the exception of perhaps a few species (e.g. Ortolan Bunting (*Emberiza hortulana*), Montagu's Harrier (*Circus pygargus*). Therefore, whilst maize expansion on most arable land has serious general detrimental impacts on bird communities, impacts on BD species are likely to be limited.

The replacement of cereals with oilseed rape can be beneficial for birds, as some species that usually nest in reed beds or other dense herbaceous vegetation have adapted to nesting in the mature crop. A study in Germany showed that Bluethroat (*Luscinia svecica*), a species that usually nests in reed beds or other dense herbaceous vegetation, has adapted to oilseed rape as a replacement nesting habitat, as it meets the species essential habitat requirements, such as shelter from predation and moist, bare soil (Berndt and Hölzel, 2012). Some commoner farmland birds, such as the Reed Bunting (*Emberiza schoeniclus*), may also nest in the crop, but these are not Annex I listed species.

Some benefits for BD birds may arise from the conversion of cereals to sugar beet fields, as these offer relatively good breeding habitat for Stone Curlews (*Burhinus oedicnemus*) and winter feeding habitat for geese, e.g. Pink-footed Goose (*Anser brachyrhynchus*) and swans, e.g. Whooper Swan (*Cygnus cygnus*) (Gillings et al, 2009).

3.2.5 Removal of residues from croplands

General biodiversity impacts

The removal of straw for bioenergy may deplete soil organic matter over the long-term, if other organic materials (e.g. cover crops, manure, or biogas digestate) are not added to compensate for the loss. It is unlikely that removal of crop residues will have any direct effects on HD or BD species, as cereal stubbles are already becoming scarce in northern European agricultural landscapes for reasons that are unrelated to bioenergy use (Bijlsma, 2013).

The impact of the removal of residues from the maintenance of landscape features such as tree lines, hedges or field margin scrub is documented together with similar residues from semi-natural environments in the next section.

3.2.6 Summary of evidence of sensitivity of EU protected habitats and species to the production of biofuel and biogas feedstocks

Figure 3-7 Risk factors associated with bioenergy from conventional food crops on EU protected species associated with agricultural mosaics and cropland

Risk factor	General effects & biodiversity impacts	Evidence of sensitivity of HD habitats	Evidence of sensitivity of BHD species	Overall sensitivity of HD habitats	Overall sensitivity of HD & BD species	Comments (incl. on link to bioenergy)
Conversion of grassland or other semi-natural habitat or fallow to bioenergy food crop.	Profound change in structure and species composition of habitat, such that no longer semi-natural and HD qualifying. Complete loss of a broad range of species.	Some MS reporting evidence of high impacts, esp for hay meadows; clear evidence from the literature	Little evidence from MS reports on plants, some for invertebrates & amphibians; wider evidence from MS reports and literature for mammals and birds	Very highly negative: all semi-natural habitats highly sensitive, i.e. destroyed	Very highly negative: vast majority of affected species lost	Strong evidence of sensitivity, and some evidence of direct impacts driven by bioenergy.
Intensification of grassland management – e.g. higher fertiliser & pesticide use, re-seeding of grassland, high stocking rates and/or switch to silage	Reduced structural and botanical diversity, invertebrate diversity and density and in turn reduced suitability as animal breeding and feeding habitat	MS reporting indicates high level pressures. Extensive evidence of major impacts	MS reporting indicates high level pressures. Extensive evidence of major impacts	Highly negative: all semi-natural habitats highly sensitive	Highly negative: most species decline and some lost	Strong evidence of sensitivity, but little evidence that grassland intensification is directly linked to bioenergy use.
Crop changes on existing arable land – cereals etc to oil-seed rape	Generally little change in fertiliser and use, pesticide, tillage and cropping season; provides breeding habitat for some birds and food resources for some pollinators, and may	Not applicable as no HD habitats on arable land	Limited evidence from MS of high impacts. Scientific evidence suggests impacts generally low	None: as no HD habitats on arable land	Low: impacts are low and variable negative / positive and only affect a few species	Generally low impacts of changes in arable crop type.

Risk factor	General effects & biodiversity impacts	Evidence of sensitivity of HD habitats	Evidence of sensitivity of BHD species	Overall sensitivity of HD habitats	Overall sensitivity of HD & BD species	Comments (incl. on link to bioenergy)
	increase crop diversity in the landscape					
Crop changes on existing arable land – cereals etc to maize	High levels of fertiliser and herbicide use, soil loss, often grown in large monocultures.	Not applicable as no HD habitats on arable land	Limited evidence from MS of high impacts. Scientific evidence indicates high impacts on a few species	None: as no HD habitats on arable land	Highly negative, but only affect a few species that use cropland	Strong evidence of sensitivity to intensification conversion to maize monoculture, partly driven by bioenergy demands, but only affects a few HD & BD species
Removal of residues from croplands	Removal of straw etc that can result in reduced food resources (e.g. grain, weed seeds, invertebrates) and organic matter returned to the soil, reducing soil fauna.	Not applicable as no HD habitats on arable land	Not reported on by MS.	None: as no HD habitats on arable land	Unlikely to have significant impacts, (although indirect impacts are expected from loss of soil organic matter), as key crop residues, notably cereal stubbles, already routinely removed	No evidence of direct impacts on HD and BD species.

3.3 Bioenergy from non-food crops or other feedstocks on agricultural land or other open land

3.3.1 Introduction

Bioenergy from non-food crops on agricultural land involves the production of the following main feedstocks:

b. Miscanthus or Reed Canary Grass (*Phalaris arundinacea*)
c. Short Rotation coppice (SRC)
d. Afforestation with fast growing trees
e. Small-scale tree planting

This involves the following risk factors that are likely to result in the main types of biodiversity impact:

- Conversion of arable cropland to Miscanthus or other non-food biomass crops or SRC
- Conversion of grassland or other semi-natural habitat to Miscanthus etc or SRC
- Afforestation of grassland or other semi-natural habitat
- Afforestation of arable cropland

The evidence for the impacts of these risk factors is detailed below. The evidence focuses on the conversion of grassland or other semi-natural habitats to non-food bioenergy crops, which is expected to be associated with the most negative impacts. In contrast, the conversion of existing arable cropland to non-food bioenergy crops and small-scale afforestation on arable land might have some positive effects depending on the landscape context and the species concerned.

Very little direct evidence was found of these crops on EU protected habitats and species, so possible effects are extrapolated based on the risk factors known to be affecting the species, and the evidence of those risk factors on similar species, and evidence of impact on food resources.

3.3.2 Conversion of arable cropland to Miscanthus or Reed Canary Grass (Phalaris arundinacea) or SRC

General biodiversity impacts

Miscanthus and Reed Canary Grass are grown as a three to five-year crop on arable fields. They are currently grown on a small surface area in the EU, as relatively small patches within arable farmland.

Short-rotation coppice (SRC) or short-rotation woody cropping (SRWC) refers to silvicultural systems designed to produce woody biomass using short harvest cycles (2-15 years), intensive silvicultural techniques, high-yielding varieties, and often coppice regeneration. The SRC species most frequently mentioned in the literature are willow (*Salix*), poplar (*Populus*), alder (*Alnus*), birch (*Betula*) and ash (*Fraxinus*) (Kretschmer et al, 2011).

Miscanthus

In general, the replacement of intensively managed arable crops with Miscanthus is expected to have a beneficial effect on biodiversity, as it provides a much less intensively managed, more permanent and more structurally diverse habitat than other arable crops. In comparison to conventionally managed arable crops, immature Miscanthus has been found to have a greater species diversity and abundance of flora, invertebrates, birds and small mammals because of the lack of disturbance and ground cover (e.g. (Bellamy et al, 2009; Bourke et al, 2014; Rowe, Street and Taylor, 2009; Semere and Slater, 2007; Smeets et al, 2014).

Some invertebrate food sources are more abundant and diverse in Miscanthus compared to arable crops. Two German studies found that earthworm species composition is more balanced in extensively managed soils under Miscanthus, compared to maize or rape fields (Gevers et al, 2011), and that Miscanthus was characterised by several rarer (because more disturbance-sensitive) carabid beetles compared to other arable crops (Dieterich et al, 2016). A study in Ireland (Bourke et al, 2014) reported positive effects (on vascular plants and solitary bees) or neutral effects (on hoverflies and bumblebees) of Miscanthus when compared with conventional crops. However, a UK study (Bellamy et al, 2009) found that Miscanthus fields provided less insect food than wheat crop plants.

It should be noted that all the above results are from immature Miscanthus plots (i.e. less than five years old), sometimes with a poorly established crop and an abundance of weeds. More established crops are likely to shade out weed flora that in turn will limit species that depend on weeds for food and/or cover. Some evidence that this may be the case comes from a study of the use of Miscanthus by birds in western Poland (Kaczmarek, Mizera and Tryjanowski, 2019). This comprised 152 ha of well-established Miscanthus fields (dense crop covering >80% of the field area) and 89 ha of poorly-established Miscanthus fields (patchy crop covering <50% of the field area) within an otherwise intensive arable farmland landscape (mainly winter cereals and oil-seed rape). Unlike the studies mentioned above, this found that the Miscanthus was used less than surrounding farmland habitats by most bird species. Whilst the authors suggest that this may be due to differences in habitat preferences between birds in Poland and Western Europe, where most of the other studies were carried out, it may also be because a large proportion of the Miscanthus was well established (although its exact age is not given in the study).

The studies of Miscanthus carried out so far also reflect the current low frequency of Miscanthus fields, and the effects of large-scale planting in the landscape (that may be necessary to supply large power plants) could result in very different impacts by reducing landscape-scale heterogeneity rather than increasing it (Bourke et al, 2014). Large-scale expansion could displace open field specialist bird species (Bellamy et al, 2009), and could therefore reduce or even reverse the positive benefits of replacing arable crops. A study in Finland showed that Skylark (*Alauda arvensis*) densities were significantly lower in reed canary grass fields than in conventionally cultivated cereal fields, because although during the early breeding season both offered similar habitat quality, the reed canary grass rapidly grows too tall and dense for field-nesting species (Vepsäläinen, 2010). Simultaneous harvest of complete areas across a landscape would also be detrimental for birds as it would remove refuges.

Short Rotation Coppice (SRC)

The impacts of SRC on Annex habitats and species depend greatly on the habitat and land use replaced by the SRC. As the trees remain in situ for at least 2 up to 15 years, the soil is not cultivated annually and SRC provides a stable habitat that is only partially shaded over by the trees, although 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. The SRC plant community is mainly defined by the species pool of ruderal species in the soil seed bank and adjacent vegetation (Baum, Weih and Bolte, 2013).

Compared to arable land, 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 for invertebrates such as bees and butterflies and a higher plant biomass than on arable land (Haughton et al, 2015). Short-rotation coppice strips can have substantial positive effects on habitat connectivity and ecosystem services on arable land, 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 (Glemnitz et al, 2013). A meta-analysis concluded that the comparative effects on biodiversity of willow and poplar SRC crops are, in general, better for biodiversity for almost all taxa (birds, mammals, butterflies, canopy invertebrates, ground beetles, rove beetles, spiders, earthworms, other soil fauna, plants) compared to other arable crops (Dauber, Jones and Stout, 2010). A review of European studies also concluded that SRC willow and poplar plantations increase farmland floral diversity compared with arable crops (Rowe, Street and Taylor, 2009).

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). 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). The trees and absence of tillage results in leaf litter accumulation on the soil surface (Baum et al, 2009), increasing the abundance of earthworms and woodlice compared to arable soils (Makeschin, 1994 cited in Baum et al, 2009; Haughton et al, 2015). 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).

SRC potentially has higher evapotranspiration rates than agricultural crops as well as some tree species, and may therefore reduce groundwater replenishment and streamflow in Mediterranean regions, although it is expected to increase water quality in comparison with agricultural crops (Dimitriou et al, 2009). However, recent measurements and modelling show that water requirements are lower than anticipated in previous reports (Fischer et al, 2018).

Evidence of impacts of Miscanthus or SRC on HD habitats on arable land

As no HD habitats occur on arable land, then no impacts will occur from conversion to these bioenergy crops.

Evidence of impacts of Miscanthus or SRC on HD species and BD birds on arable land

We found no evidence of impacts on HD species or BD birds, and this may be due to the relatively recent introduction of the crops to farmland in Europe and their small areas, and related lack of studies on their impacts. However, although the crops appear to provide some biodiversity benefits for generalist species it seems unlikely that habitat structure, composition and situation within arable landscape will be suitable for EU protected species, which tend to be specialists. Thus, it is also probably likely that the lack of evidence is also to some extent a true reflection of their limited beneficial value to such species.

Similarly, the lack of evidence of negative impacts probably reflects the lack of EU protected species in the agricultural landscape, and thus the low risk of detrimental impacts. However, significant impacts could arise if these crops were introduced to low intensity dry cereal areas such as in Spain and Portugal, which as discussed above, are of high conservation value, especially for BD birds.

In conclusion, SRC and Miscathus can provide vegetation cover that is under lower management intensity that on conventional arable land, and may therefore provide a refuge for a wide range of species; but impacts on EU protected species are uncertain (especially regarding Miscanthus due to the low number of relevant studies, which reflects the current low frequency of Miscanthus fields). Furthermore, the impacts of large-scale planting of SRC and Miscanthus in the landscape (that may be necessary to supply large power plants) could result in very different impacts by reducing landscape-scale heterogeneity rather than increasing it, and by replacing low intensity arable areas and fallow of high conservation value.

3.3.3 Conversion of grassland or other semi-natural habitat to Miscanthus or Reed Canary Grass (Phalaris arundinacea) or SRC

General biodiversity impacts

The conversion of semi-natural grasslands or other semi-natural habitats to Miscanthus or Reed Canary Grass is likely to be associated with major negative effects on typical grassland and scrub species, similarly to the impacts of grassland conversion to conventional crops discussed in the previous section (although without fertiliser and pesticide runoff or drift effects). The process requires the complete removal of the semi-natural vegetation, followed by tillage and soil preparation and fertilisation. In general, however, this practice is currently relatively rare, as both crops require fertile tilled arable soil in order to produce an economically viable crop, and so are more likely to be planted on arable land or improved grassland.

Some SRC is being planted on grassland rather than arable land; for example, in Ireland the area of dedicated bioenergy crops (mainly SRC but also Miscanthus) has expanded by replacing permanent grassland[78]. Replacing grassland with SRCs may have positive or negative effects, depending on the intensity of the grassland use (Dauber, Jones and Stout, 2010). For example, a study found that earthworms are not more abundant in SRC than under grassland (Dauber, Jones and Stout, 2010). Compared to woodland habitats, SRC biodiversity benefits are likely to be equal or decrease and depend upon SRC type (Dauber, Jones and Stout, 2010). 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 woodland. However, created bird nesting habitats will be destroyed by the harvesting.

Overall, the habitat is unlikely to be suitable for HD species under their typical intensive management conditions. The BIOSCORE model predicted that large-scale short-rotation coppice production on open agricultural land and abandoned land would have a negative net effect on all taxonomic groups except plants (Louette et al, 2010). The model predicted negative effects from the second-generation bioenergy scenario (SRC expansion) for 40% of reptiles in all regions, around 25% of butterflies in the Atlantic, Continental and Mediterranean regions, and for 25% of species in the Boreal region. It also predicted that more than 35% of bird species in the Mediterranean region would be negatively affected in the scenario (Louette et al, 2010).

Evidence of impacts of Miscanthus or SRC on HD habitats

Although no specific studies were found on the subject, it is evident that all HD habitats would be destroyed by their conversion to bioenergy crops.

Evidence of impacts of Miscanthus or SRC on HD species on non-arable land

The HD vascular plants, bryophytes and lichens characteristic of forests generally have a low colonisation ability, and so are unlikely to have time to develop in SRC because of the short rotation period (McKay, 2011). The HD species of grasslands and croplands are adapted to open land and not able to survive in the shady conditions of established SRC. 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 the HD listed saproxylic species is expected to be low (McKay, 2011), although densities of more ubiquitous saproxylic forest arthropods have been shown to increase as SRC ages (Glemnitz et al, 2013). There is evidence that foraging 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 the HD butterflies.

Evidence of impacts of Miscanthus or SRC on BD birds on non-arable land

The bird community of SRC include some open land species initially, and then is closer to that of scrub than forest (Dauber, Jones and Stout, 2010). A review of European studies concluded

[78] A Irish government report in 2017 reported 3,353 ha of bioenergy crops (Miscanthus, short rotation coppice etc) established under the bioenergy scheme, available at https://www.dccae.gov.ie/documents/NREAP%20Fourth%20Progress%20Report.pdf

that SRC willow and poplar plantations increase the abundance of common birds associated with scrub and woodland habitats compared with arable crops, and also some rarer species, but not species of European conservation concern (Rowe, Street and Taylor, 2009). Anderson and Fergusson (2006) concluded that, in comparison with conventional arable crops, large-scale short-rotation willow coppice plantations can provide benefits for some taxonomic groups, e.g. bird species typical of rank herbaceous vegetation, scrub and young woodland (Sage, Cunningham and Boatman, 2006). However, if more marginal, less intensively managed land is replaced by biomass crops, then effects on overall biodiversity are more likely to be negative (Anderson and Fergusson, 2006). It is also important to note that, as well as the loss of habitat area, the presence of the SRC in the landscape may reduce the suitability of nearby areas for breeding habitat for open farmland specialists, including BD species such as the Stone Curlew (*Burhinus oedicnemus*) (Rowe, Street and Taylor, 2009).

3.3.4 Afforestation of grassland or other semi-natural habitat

General biodiversity impacts

In some circumstances afforestation on open ground 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, 2008b). However, the afforestation of semi-natural habitats with their unique and/or rich biodiversity is more likely to result in a substantial net loss of nature conservation value including EU protected habitats and species. Furthermore, this type of land is more likely to be afforested because of the potential economic benefits, particularly where agricultural management is being abandoned.

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, 2008b).

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. It is also important to note that the impacts of afforestation are influenced by forest management practices. In southern Sweden, hybrid aspen stands had higher bird species richness and abundance as well as a distinct community composition compared to young spruce stands because they are fenced against deer grazing and therefore have the greater structural and tree species complexity (Lindbladh et al, 2014).

Evidence of impacts of afforestation on HD habitats

14 Member States reported under the EU Habitats Directive for 2007-2012 that afforestation on open land was a high-level pressure on 38 different non-forest Annex I habitat types, and 7 of those reported high level pressure from afforestation with non-native tree species on 8 non-forest Annex I habitat types (Figure 3-8). Two heath/shrub non-forest Annex I habitat types were reported as under high pressure from forest planting, in actual fact afforestation on open ground. A number of Member States also reported afforestation on open ground as a high level pressure on forest habitats, but it was not possible in this study to investigate whther this is a genuine pressure on open areas within these habitats or if it is a mistaken reporting and should actually be forest replanting (see 3.4.7). Atlantic wet heathlands (HD habitat 4020) are also mentioned in a study as under high pressure from afforestation with Eucalyptus and pines in Galicia, Spain (Muñoz-Barcia et al, 2019). A survey of experts in the Member States that made significant use of the RDP afforestation measure up to 2013 reported 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 2004[79] in the Czech Republic, Estonia, Hungary, and Lithuania, and since 1992[80] in the UK, Ireland and Spain (Elbersen et al, 2014). In contrast, experts from those Member States that targeted afforestation to arable and other cropped land (i.e. not to marginal land) reported no negative effects (Denmark, Poland, Romania)[81] (Elbersen et al, 2014).

Afforestation on scrubland or semi-natural grassland generally results in a decrease in plant species richness compared to the unforested habitat (Bremer and Farley, 2010), as for example was found to have occurred five years after afforestation of wet grassland in Ireland with *Picea sitchensis* (Buscardo et al, 2008), 25 years after afforestation of Mediterranean garrigue scrub in Italy with *Pinus halepensis* (Salvatore, La Mantia and Rühl, 2012), and 30-40 years after afforestation of Mediterranean semi-arid scrub in Spain with pines (Gómez-Aparicio et al, 2009).

Two non-indigenous species are being used for afforestation and short-rotation coppice - 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 pressure to a number of HD habitats (Baranska, 2015; Kleinbauer et al, 2010; WWF Hungary, 2011) and controlling *Robinia* plantations is mentioned as a management measure in many Natura 2000 sites (Braun, Schindler and Essl, 2016).

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79 Afforestation of non-agricultural land under Article 45 of Regulation 1698/2005

80 Afforestation under Regulation 2080/92, Regulation (EC) No 1257/1999, and Article 45 of Council Regulation (EC) No 1698/2005

81 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).

Figure 3-8 Habitats Directive Annex I habitats reported as under high pressure from artificial planting of non-native trees on open ground

Key to biogeographical regions: ALP Alpine; ATL Atlantic; BLS Black Sea; BOR Boreal; CON Continental; MAC Macaronesian; MED Mediterranean; PAN Pannonian; STE Steppic.
MS = Member State H = high M = medium L = low
Evidence of sensitivity according to the literature – see text for related reference sources.

Habitat code	Habitat name	MS reporting high pressure	Evidence of sensitivity in the literature
1240	Vegetated sea cliffs of the Mediterranean coasts with endemic Limonium spp.	MT (MED)	
6110	Rupicolous calcareous or basophilic grasslands of the Alysso-Sedion albi	SK (ALP,PAN)	
6220	Pseudo-steppe with grasses and annuals of the Thero-Brachypodietea	ES (ATL,MED)	
6230	Species-rich Nardus grasslands, on siliceous substrates in mountain areas (and submountain areas, in Continental Europe)	BE (CON), IT (MED)	
6510	Lowland hay meadows (Alopecurus pratensis, Sanguisorba officinalis)	IT (CON)	
7130	Blanket bogs (* if active bog)	IE (ATL)	
7150	Depressions on peat substrates of the Rhynchosporion	IE (ATL)	
7230	Alkaline fens	BE (CON)	
9120	Atlantic acidophilous beech forests with Ilex and sometimes also Taxus in the shrublayer (Quercion robori-petraeae or Ilici-Fagenion)	IT (CON)	
91D0	Bog woodland	PT (MAC)	
9190	Old acidophilous oak woods with Quercus robur on sandy plains	IT (CON)	
9160	Sub-Atlantic and medio-European oak or oak-hornbeam forests of the Carpinion betuli	IT (CON,MED)	
9220	Apennine beech forests with Abies alba and beech forests with Abies nebrodensis	IT (MED)	

Figure 3-9 Habitats Directive Annex I non-forest habitats reported as under high pressure from forest replanting

Habitat code	Habitat name	MS reporting high pressure	Evidence of sensitivity in the literature
2180	Wooded dunes of the Atlantic, Continental and Boreal region	BG (BLS), LV (BOR)	
5120	Mountain Cytisus purgans formations	ES (ALP)	
4030	European dry heaths	BE (CON)	

Evidence of impacts of afforestation on HD species

Afforestation is reported as a high-level pressure on a large number of HD species in nearly half of Member States. 16 Member States reported 80 HD species as under high pressure from afforestation on open ground, whether with native or non-native tree species, during the 2007-2012 period. France reported 42 HD species as under high pressure from afforestation, Bulgaria reported 15 HD species, Spain 11 HD species, Hungary 10 HD species, and the Czech Republic 7 HD species.

There is evidence that Eucalyptus plantations negatively affect aquatic biodiversity, though the studies do not directly refer to EU protected habitats and species. A study compared fish assemblages in streams in eucalypt plantations with protected and managed riparian buffer strips (as prescribed by forest certification scheme), riparian forests with unmanaged riparian zones, and native oak forests (Oliveira, Fernandes and Ferreira, 2016). Compared to the native oak forests, the riparian buffers in the certified Eucalyptus plantations generally had lower potential cover for fish and tended to support a riparian vegetation that was a little more fragmented, with a lower abundance of native invertivores. The unmanaged zones had disturbed riparian vegetation, degraded, and modified channels, and degraded stream habitats, and substantially altered fish assemblages. The authors conclude that the protected riparian zones in the certified forests mitigate the effects of eucalypt forestry on fish assemblages, but that protecting their biological integrity would require restoration of the native vegetation, removal of alien plants, and improvement of the stream habitat. A parallel study (Ferreira et al, 2015) found no impacts on microbial decomposition of eucalyptus leaf litter in Portuguese and Spanish streams, likely due to high functional redundancy among microbes. However, the macro-invertebrate decomposer communities were slower to colonise high quality (alder leaf) litter in the eucalyptus streams compared to streams in native oak forest. The Portuguese eucalyptus plantations lacking deciduous riparian corridors had fewer macro-invertebrates than the Spanish ones which did. Another study (Cordero–Rivera, Martínez Álvarez and Álvarez, 2017) that compared 16 streams in Spain found lower macro-invertebrate species richness in the Eucalyptus dominated streams and a higher likelihood that these streams dry up in summer, which has a marked detrimental effect on macro-invertebrate communities.

Arntzen (2015) argued that the Golden-Striped Salamander (*Chioglossa lusitanica*) was negatively affected by the presence of *Eucalyptus* sp. in two areas of one mountain range in northwestern Portugal. The main reasons for the decrease were attributed to the reduction of space for native wildlife and the decrease of the water table in adjacent streams. A laboratory experiment showed that eucalypt tree chemicals reduced the ability of male newts (*Lissotriton helveticus*) to detect female chemicals compared to newts in water soaked in native oak litter, and they were also less able to detect conspecific alarm cues signalling predatory events (Iglesias-Carrasco et al, 2017).

A study found lower *Pipistrellus* bat activity levels in all Eucalyptus plantation stands than native montado habitat (Cruz et al, 2016), though increasing age can make a difference as bat activity and species richness were higher in eucalypt plantations with complex understorey vegetation structure (from the ground level up to 3 m high), and proximity to water points.

Figure 3-10 Habitats Directive species reported as under high pressure from forest planting on open ground (including planting of native and non-native species)

Key to biogeographical regions: ALP Alpine; ATL Atlantic; BLS Black Sea; BOR Boreal; CON Continental; MAC Macaronesian; MED Mediterranean; PAN Pannonian; STE Steppic.
MS = Member State H = high M = medium L = low
Evidence of sensitivity according to the literature – see text for related reference sources.

Taxonomic group	Species name	Member State reporting high pressure	Evidence in the literature
MAMMALS	*Bison bonasus*	PL (CON)	
MAMMALS	*Barbastella barbastellus*	SE (BOR)	

MAMMALS	*Felis silvestris*	PL (ALP, CON)	
MAMMALS	*Mesocricetus newtoni*	BG (BLS, CON)	
MAMMALS	*Microtus tatricus*	PL (ALP)	
MAMMALS	*Muscardinus avellanarius*	FR (ALP)	
MAMMALS	*Plecotus macrobullaris, Rhinolophus euryale*	FR (MED)	
REPTILES	*Ablepharus kitaibelii, Coluber caspius, Elaphe longissimi, Podarcis muralis, Podarcis taurica*	HU (PAN)	
REPTILES	*Coronella austriaca*	SE (BOR)	
REPTILES	*Emys orbicularis*	LT (BOR)	
REPTILES	*Lacerta agilis*	BE (CON), DE (ATL), EE (BOR), FR (ATL), HU (PAN), LT (BOR), SE (CON)	
REPTILES	*Lacerta viridis*	DE (CON), HU (PAN)	
REPTILES	*Vipera ursinii*	FR (ALP, MED)	
AMPHIBIANS	*Bufo calamita*	EE (BOR)	
AMPHIBIANS	*Pelobates cultripes*	FR (MED)	
AMPHIBIANS	*Pelobates fuscus*	FR (CON), SE (CON)	
AMPHIBIANS	*Triturus cristatus*	FR (ALP)	
MOLLUSC	*Vertigo angustior*	FR (ALP, CON), LT (BOR)	
ARTHROPODS (saproxylic beetle)	*Lucanus cervus*	CZ (CON, PAN), RO (CON, PAN, ALP, STE)	
ARTHROPODS (butterflies)	*Coenonympha hero*	FR (CON), SE (BOR), AT (ALP)	
ARTHROPODS (butterflies)	*Coenonympha oedippus*	FR (ATL)	
ARTHROPODS (butterflies)	*Colias myrmidone, Polyommatus eroides*	PL (CON)	
ARTHROPODS (butterflies)	*Erebia sudetica*	CZ (CON)	
ARTHROPODS (butterflies)	*Euphydryas aurinia*	FR (ATL), LT (BOR), SE (BOR)	
ARTHROPODS (butterflies)	*Lycaena helle*	FR (ALP, CON)	
ARTHROPODS (butterflies)	*Maculinea arion*	FR (ALP, MED)	
ARTHROPODS (butterflies)	*Maculinea teleius*	FR (ATL), LT (BOR)	
ARTHROPODS (butterflies)	*Parnassius apollo*	PL (ALP)	
ARTHROPODS (butterflies)	*Plebicula golgus*	ES (MED)	
NON-VASCULAR PLANTS	*Bruchia vogesiaca*	ES (ATL), ES (MED)	
NON-VASCULAR PLANTS	*Diphasiastrum alpinum*	FR (ALP)	
NON-VASCULAR PLANTS	*Huperzia selago*	FR (ALP, MED)	
NON-VASCULAR PLANTS	*Sphagnum spp.*	FR (ATL)	
VASCULAR PLANTS	*Aquilegia bertolonii*	FR (ALP)	
VASCULAR PLANTS	*Arnica montana*	AT (CON), FR (ALP, CON)	
VASCULAR PLANTS	*Centaurea immanuelis-loewii*	BG (CON)	
VASCULAR PLANTS	*Cypripedium calceolus*	FR (CON, MED)	
VASCULAR PLANTS	*Dracocephalum austriacum*	FR (MED)	
VASCULAR PLANTS	*Euphorbia transtagana*	PT (MED)	
VASCULAR PLANTS	*Gentiana lutea, Gladiolus palustris*	FR (CON)	
VASCULAR PLANTS	*Himantoglossum adriaticum*	AT (ALP)	
VASCULAR PLANTS	*Iris boissieri*	ES (ATL, MED)	
VASCULAR PLANTS	*Jungermannia handelii*	ES (ATL)	
VASCULAR PLANTS	*Leuzea longifolia*	PT (MED)	
VASCULAR PLANTS	*Lycopodium spp, Lycopodiella inundata*	FR (ALP, ATL, CON, MED)	
VASCULAR PLANTS	*Narcissus cyclamineus*	PT (ATL, MED)	
VASCULAR PLANTS	*Onopordum carduelium*	ES (MAC)	
VASCULAR PLANTS	*Spiranthes aestivalis, Thorella verticillatinundata*	FR (ATL)	
VASCULAR PLANTS	*Teucrium lepicephalum*	ES (MED)	

Evidence of impacts of afforestation on BD 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 as indicated in Member State Article 17 reports (Figure 3-11) and the literature. 6 Member States reported reported high pressure from afforestation with non-native tree species on 6 BD birds (Figure 3-11) during 2007 to 2012.

In some 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 (Voríšek et al, 2010). 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 Lark (*Melanocorypha calandra*) and Short-toed Lark (*Calandrella brachydactyla*), but not of Little Bustard (*Tetrax tetrax tetrax*) and Tawny Pipit (*Anthus campestris*) (Reino et al, 2009). The steppe birds tended to reach the highest species richness and abundances in large arable patches. Declines in Red-backed Shrike *(Lanius collurio)* have been attributed to the loss and fragmentation of habitat resulting from afforestation and agricultural intensification, and the increased use of pesticides causing loss of food resources (Yosef and Christie, 2012).

Figure 3-11 Birds Directive Annex I species reported as under high pressure from artificial planting of non-native trees on open ground

Key to biogeographical regions: ALP Alpine; ATL Atlantic; BLS Black Sea; BOR Boreal; CON Continental; MAC Macaronesian; MED Mediterranean; PAN Pannonian; STE Steppic.
MS = Member State H = high M = medium L = low
Evidence of sensitivity according to the literature – see text for related reference sources.

Species name	MS reporting high pressure from afforestation	Evidence of sensitivity in the literature
Aegypius monachus	BG	
Calandrella brachydactyla		PT
Coracias garrulus	RO	
Hieraaetus pennatus	PT	
Lanius collurio		Europe
Lullula arborea	BE	
Melanocorypha calandra		PT
Otis tarda		Europe
Philomachus pugnax	LT	
Tringa glareola	UK	

3.3.5 Afforestation of arable cropland

General biodiversity impacts

Afforestation refers to tree planting on arable land to produce wood generally in a 15 to 30 year period. Other forms of afforestation on arable land include agroforestry, planting of tree lines, hedgerows, and small woody patches. In some circumstances small scale afforestation on arable land can be expected to have positive impacts on biodiversity, by 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, 2008b). However, there is usually no link between these actions and bioenergy use.

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.

Evidence of impacts on HD habitats

As no HD habitats occur on arable land, then no impacts will occur from afforestation.

Evidence of impacts of afforestation on arable land on HD species

Afforestation of arable land can provide a replacement habitat for some HD listed species associated with woodland, but only after several decades and only if the afforestation is close to existing natural habitat. Afforested arable land is also likely to be dominated by generalist plant communities that have grown up from the seed bank or colonised from nearby farmland habitats. 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 to 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).

A UK study found higher bat foraging activity and relative abundance associated with small and isolated woodland fragments, and in sparsely wooded landscapes in agricultural landscapes, suggesting a more intensive use of woodland in landscapes where this habitat is scarce, and possible benefits for bats of afforestation on farmland (Fuentes-Montemayor et al, 2013).

Evidence of impacts of afforestation on arable land on BD bird species

It is clear that the afforestation of arable cropland will lead to the loss of most open habitat species, as is the case with grass and habitats described above. However, as most arable land in the EU supports very few BD birds, afforestation impacts on the species will be limited. The exception will again be in relation to extensive dry arable farming systems such as those in the Iberian Peninsula, where afforestation has undoubtedly led to impacts on open field species such as bustards, sand grouse and larks. Thus, amongst other pressures, the Great Bustard (*Otis tarda*) is considered to be impacted by afforestation and increasing development of irrigation schemes, roads, power-lines, fencing and mechanised farming (Nagy, 2018). Afforestation can also provide nesting sites and cover for predators, which can then lead to high rates of mortality of nearby ground nesting species.

A few open farmland species that require trees or forest habitats for nesting may benefit from small-scale tree planting or limited afforestation in largely treeless landscapes. For example, The EU Species Action Plan for the Red-footed Falcon (*Falco vespertinus*) (Palatitz et al, 2010) notes that the species is highly dependent on nesting habitat in tree groups occupied by Rooks (*Corvus frugilegus*) within agricultural areas. A restoration programme in Hungary is therefore planting small tree patches in suitable agricultural areas with semi-natural grassland.

3.3.6 Removal of residues from semi-natural habitats

General biodiversity impacts

The use of residues and cuttings from extensive management of semi-natural habitats for bioenergy has been promoted as a way to restore and maintain semi-natural grasslands in Estonia (Esko and Holm, 2017; Heinsoo et al, 2010), Germany and other countries. There are also many cases where the use of residues from the conservation management of semi-natural habitats (e.g. coppiced woodland) for bioenergy subsidises the management costs, or in some cases provides a profit. In initiatives that are successful for biodiversity, the driving factor is the nature conservation requirements rather than the biomass production, which is rather low in terms of actual energy generation. In fact, the initiatives must build in sufficient governing mechanisms to ensure that biomass production is not intensified to increase revenues, thereby destroying the biodiversity benefits.

3.3.7 Summary of evidence of sensitivity of EU protected habitats and species to the production of bioenergy crops

Figure 3-12 Risk factors associated with bioenergy crops on EU protected species associated with agricultural mosaics and cropland

Risk factor	**General effects & biodiversity impacts**	**Evidence of sensitivity of HD habitats**	**Evidence of sensitivity of BHD species**	**Overall sensitivity of HD habitats**	**Overall sensitivity of BHD species**	**Comments (incl. on link to bioenergy)**
Conversion of arable cropland to Miscanthus or other non-food biomass crops or SRC	Created habitats likely to be of low biodiversity value, but may increase habitat diversity and/or create habitat features that can help link up other habitats across the landscape.	Not applicable as no HD habitats on arable land	Potential positive impact for common farmland species due to refuge effect of small-scale cultivation, but impacts of large-scale expansion still uncertain.	None: as no HD habitats on arable land	Low-level impacts which are likely to be variable depending on species involved for a relatively small number of cropland species (mainly birds)	Overall impacts are currently still low as cropping area is still small scale and restricted to a few regions (mainly grown for bioenergy but also for other uses).
Conversion of grassland or other semi-natural habitat to Miscanthus or other non-food biomass crops or SRC	Profound change in habitat type and associated species communities – replacement of open cropland species with generalist species of shrubland	No specific studies were found on the subject, but it is evident that all HD habitats would be destroyed	No MS reporting information, and limited scientific information as these are relatively new bioenergy crops, but detrimental impacts inferred from species requirements	Very highly negative: all semi-natural habitats highly sensitive, i.e. destroyed	Very highly negative - most associated species will be lost	Overall impacts currently low as very little conversion to these crops has taken place, but impacts of possible future large-scale conversion likely to be highly negative
Afforestation of grassland or other semi-natural habitat	Profound change in habitat type and associated species communities – loss of open diverse habitat and specialist species and increase in forest generalists	14 MS reported high pressure on 38 non-forest habitats in 66 regions.	MS reports of high pressure on 6 birds. Scientific evidence of impacts on plants and birds.	Very highly negative: All semi-natural habitats highly sensitive, i.e.. destroyed	Very highly negative - most associated species will be lost	Some biodiversity benefits possible in certain situations – eg small patches to link up fragmented habitats, or on improved grasslands

Risk factor	General effects & biodiversity impacts	Evidence of sensitivity of HD habitats	Evidence of sensitivity of BHD species	Overall sensitivity of HD habitats	Overall sensitivity of BHD species	Comments (incl. on link to bioenergy)
Afforestation of cropland	As above for grassland, but impacts are less due to the lower biodiversity of most cropland landscapes	Not applicable as no HD habitats on arable land	High sensitivity of bird specialists of open low intensity farmland, such as Otis tarda and Melanocorypha calandra	None: as no HD habitats on arable land	Very high negative impacts but on a small number of cropland species (mainly birds)	Afforestation of low intensity arable land negative for specialist bird species. Small scale afforestation with locally native deciduous tree species can provide benefits but mostly not related to bioenergy demand.
Removal of residues from semi-natural habitats	If carried out in accordance with conservation objectives – can help counteract succession (e.g. resulting from under-grazing) and/or vegetation growth and changes resulting from eutrophication. Intensive removal of vegetation would be damaging.	Some cases of successful initiatives where habitat management or restoration is supported by use of residues for bioenergy – but requires careful safeguards to prevent intensification of biomass removal	Some cases of successful initiatives where species habitat management or restoration is supported by use of residues for bioenergy – but requires careful safeguards to prevent intensification of biomass removal	Moderate: many habitats are dependent on regular low intensity biomass removal but highly sensitive to intensification of biomass removal (see grassland intensification summary)	Moderate: as for habitats	Some cases of successful initiatives where habitat restoration is supported by use of residues for bioenergy – but requires careful safeguards to prevent intensification of biomass removal

3.4 Impact of removal of forest biomass on biodiversity

3.4.1 Introduction

The effects of biomass removal for bioenergy production in forests covered in this literature review encompass all risk factors associated with the direct use of the two main sources of biomass: primary round wood production, and the extracted residues of harvesting and forestry management practices. It also considers the indirect effects of forest biomass demand for bioenergy production, which includes increases in the area of forest under exploitation for timber and other non-bioenergy resources, and/or the intensification of forest management to increase the production of such forest resources.

The identified risk factors take into account the general impacts of forest exploitation on biodiversity. These very much depend on the mix of tree species, and the cultivation, harvest and stand management approaches within forests. Although forest management is site-specific and depends on the site objectives, these factors can be classified according to the typology developed by Duncker et al (2012), which is based on an increasing degree of manipulation and management intensity (Figure 3-13).

Figure 3-13 Definition of forest management types from an ecosystem services perspective

Management alternatives			Description and management objective	Tree species Rules	Site management and cultivation approach	Harvest and stand management approach
<-------------- Increasing degree of manipulation and management intensity <--------------	Extensive forestry	Unmanaged forest nature	No management. Natural disturbances and succession drives development. Reference for authenticity and biodiversity refuge.	Natural	Not applicable	Not applicable
		Close-to-nature forestry	Stand management that mirrors natural processes as a guiding principle. Economic outturn is important but must occur within the frame of this principle.	Natural or adapted	Mostly natural regeneration without soil tillage. None or only exceptional chemical or physical site manipulations.	Thinnings are extensive and selective. Final harvesting often according to target diameter
		Combined objective forestry	An alternative defined by man characterised by inclusion of several considerations and goals, e.g. social, environmental and economic.	Often natural or adapted.	Cultivation might be artificial after site and soil preparations. Fertilisation and use of pesticides and other physical manipulation are limited.	Thinnings and stand regulation are often performed. Rotation lengths often increased for environment and social reasons.
	Intensive forestry	Intensive even-aged forestry	The main objective of intensive even-aged forestry is to produce timber. If ecological aims can be achieved without much loss of revenue, they are	Optimal according to production purpose.	No restrictions besides general national legislation or guidelines. Site preparation used to improve establishment success and fertilisation to increase growth rates. Planting/seed material can	No restrictions besides general national legislation or guidelines. Rotation length depends mainly on the economic return and is normally similar to or shorter than the age of Maximum Mean Annual Increment.

Management alternatives			Description and management objective	Tree species Rules	Site management and cultivation approach	Harvest and stand management approach
			normally incorporated.		be genetically improved, but not modified.	
		Short-rotation forestry	Focus is only on production of fibres typically for energy or pulp. Often short rotation coppice (on agricultural land). Could be called lignoculture.	No restrictions. Tree species selection depends mainly on economic returns.	Planting material can be genetically improved and/or modified. Sites are cultivated and can be drained or irrigated. Fertiliser/lime applied to enhance growth. Chemicals used to control pests, weeds and diseases.	Rotation length depends only on economic returns (≤20 yrs.). Final clearcut harvesting combined with removal of all woody residues if suitable markets exist.

Taken from (Hart et al, 2013)- Adapted from (Duncker, Spiecker and Tojic, 2007) and (Duncker et al, 2012).

The impact of forestry management on biodiversity depends on both the original forest type that is replaced and the new forest management. Forestry practices can have both positive and negative effects on biodiversity, depending on the type of management and taxonomic group. However, a general negative effect on biodiversity has been identified with increasing management (Paillet et al, 2010). Furthermore, the requirements of specialist species of conservation concern may not follow general biodiversity patterns. The following typical effects were identified from the literature:

Positive effects:

- The creation of small gaps in closed forest canopies through thinning or selective felling can increase overall species richness (eg of plants, lichen and bryophyte species), due to the creation of patches of sunny, dry exposed conditions (Nordén et al, 2012; Paltto, Nordén and Götmark, 2008).
- Some forest types were created by traditional management systems that maintain an open forest structure, including coppicing, pollarding, and forest pastures or meadows[82]. These forests provide suitable habitat for invertebrates (such as butterflies) and plant communities that require an open canopy, and also host different bird communities to closed forest.

Negative effects:

- Selected felling of old trees has a negative impact on various taxonomic groups, including bryophytes, lichens, fungi, saproxylic beetles, molluscs and birds (Bouget et al, 2014; Brin et al, 2011; Brunialti et al, 2010; Cuttelod, Seddon and Neubert, 2011; Fritz and Brunet, 2010; Gutowski et al, 2014; Horák, Vávrová and Chobot, 2010; Jonsell, Hansson and Wedmo, 2007; Kostanjsek et al, 2018; Lassauce, Lieutier and Bouget, 2012; Moning and Müller, 2009; Paillet et al, 2010; Wesołowski and Martin, 2018).

[82] Annex I habitats of this type include Fennoscandian wooded pastures (H9070), sub-Atlantic and medio-European oak or oak-hornbeam forests of the Carpinion-betuli (H9160), Galio-Carpinetum oak-hornbeam forests (H9170), Castanea sativa woods (H9260), Quercus suber forests (H9330)

- Clear cutting destroys entire habitats and species communities (eg epiphytic species) found in forests since all standing trees are removed (Dynesius, 2015; Dynesius and Hylander, 2007; Knorn et al, 2013; Sahlin, 2010; Zaghi, 2008).
- Stump and whole tree harvesting has a significant negative impact on saproxylic invertebrate diversity (Brin et al, 2013; Horák, Vávrová and Chobot, 2010; Jonsell and Hansson, 2011; Jonsell and Schroeder, 2014; Russo, Cistrone and Garonna, 2011b; Victorsson and Jonsell, 2013a, b).
- Thinning of small and intermediate sized successional trees may have an impact on biodiversity. For instance, it may increase the extinction rate of specialist epixylic bryophytes that depend on closed canopy (Paltto, Nordén and Götmark, 2008).
- Small and large diameter dead wood removal has a negative impact on species of conservation concern in forests, including saproxylic bryophytes, lichens, fungi and invertebrates (Bergmeier, Petermann and Schröder, 2010; Brin et al, 2011; Humphrey et al, 2002; Jonsell, Hansson and Wedmo, 2007; Lassauce et al, 2011; Moning and Müller, 2009).
- Forestry operations can lead to disturbance of local fauna e.g. machinery noise (Benítez-López, Alkemade and Verweij, 2010; Capitani et al, 2006; Gurarie et al, 2011; Güthlin et al, 2011; Kaartinen, Kojola and Colpaert, 2005).

Based on these effects and the forest management typology, the following list of risk factors from the direct and indirect effects of bioenergy production from forest biomass were identified and are further described below. Although they are often interrelated in practice, they reflect the general trend towards increasing intensity of biomass extraction for bioenergy production. Risk factors to be investigated on an increasing scale of intensification:

- Selective logging of mature trees in extensively managed forests (to produce roundwood for biomass or other purposes).
- Thinning of small and intermediate sized successional trees (to increase roundwood productivity for biomass or other purposes, and/or also potentially producing residues for bioenergy).
- Removal of dead wood e.g. to reduce disease and fire risk (to increase roundwood productivity for biomass or other purposes, and/or also potentially producing residues for bioenergy).
- Clear cutting of trees to increase production efficiency (to increase roundwood productivity for biomass or other purposes).
- Stump removal and whole-tree harvesting (to increase harvesting efficiency and residues or bioenergy).
- Tree planting and use of pesticides and fertilisers (to increase roundwood productivity for biomass or other purposes).
- Plantation forestry with dominance of non-native or non-site-typical tree species (to increase roundwood productivity and efficiency for biomass or other purposes).
- Construction of forest roads and tracks to support forest management and harvesting.

Within Natura 2000 sites, there is the requirement that forest management is in accordance with the conservation measures identified for the habitats and species of EU interest present in that site, so even though Natura 2000 designation does not preclude continued forestry activites and wood extraction, this should take place in a way that maintains or enhances the conservation status of the habitats and species of EU interest. These forestry-related pressures can therefore be expected to mainly apply to the habitats and species of EU interest that are outside the Natura 2000 network. However, not all forests inside the Natura 2000 network are being managed in full accordance with their conservation objectives, as recognised in the recent study published by the European Forest institute (Sotirov, 2017). In addition, there is often still a lack of clarity as to how the conservation measures translate into specific forest management guidance that forest managers can implement[83]. Outside the network, whilst some managers may decide not to intensify extraction in order to remain within limits set by a sustainable forest management plan or forest certification, these instruments do not necessarily preclude more intensive extraction, and they do not cover all forest areas (see main report for more discussion of policy measures).

3.4.2 Selective logging of extensively managed forests

General biodiversity impacts

Logging is the process in which trees are felled, particularly for timber harvest. Selective logging of forests is a management method that involves the felling of only a number of selected trees that are chosen to be in the right condition for harvesting. Other trees and tree groups are left standing, either because they have no value for logging or for further growth, and therefore the method maintains a high proportion of forest cover. It is not widely used in the EU for industrial forestry activities[84], however it can be associated with local firewood markets and forests managed for multiple objectives.Selective logging can retain an uneven aged forest structure that resembles the natural forest condition by mimicking natural disturbances, and is therefore expected to have less impact on most forest habitats and species than clear cuting. For example, a study in Sweden comparing selective felling in uneven-aged forests and clear-cut forests with uneven-aged forests without forestry activity, and found that the overall abundance, species richness and composition of beetles (Coleoptera) was not significantly affected by the selective felling regime, whilst the clear cut and thinned even-aged forests had different beetle assemblages even 50 years into the rotation (Joelsson et al, 2017). However, the authors point out that that their conclusions cannot be applied to the beetle species on the Swedish red list, as their sample size was too low to draw conclusions about the impacts of selective felling. The impact on most species depends on the relative intensity of tree felling compared to the forest left standing, and to what degree the selective felling targets trees associated with dead or dying wood or other features such as tree holes.However, the initiation of selective logging in old growth forest habitats has the potential to affect highly sensitive specialised species.

Old-growth forest areas host many typical forest species with a low ability to migrate or recolonise new forest patches (e.g. many invertebrate groups, some plants and mosses), and

[83] However, not all forests inside the Natura 2000 network are being managed in full accordance with their conservation objectives, as recognised in the recent study published by the European Forest institute (Sotirov et al 2017).
[84] http://ec.europa.eu/environment/nature/natura2000/management/gp/forest_intro.html

they are of particular importance for the protection of forest HD habitat types and species. Managed forests are generally poorer in dead wood in advanced stages of decomposition. .

None of the pressure codes used by Member States to report under the Birds and Habitats Directives clearly relate to the act of selective logging alone. Therefore, the analysis below only draws on evidence of the impacts of this activity from the scientific literature.

Evidence of impacts of selective logging on HD habitats

Selective logging results in changes in tree age structure, vertical stratification and tree species composition (Bogunić et al, 2005; Paillet et al, 2010; Standovár et al, 2006), affecting the availability of micro-habitats in the forest (Berg et al, 1994; Christensen et al, 2005; Gibb et al, 2005). Currently, the EU still has around 1.4 M ha of unmanaged old growth forest, mostly in the boreal biogeographical region and the alpine regions (as well as the remaining Laurisilva forest on Madeira) (Sabatini et al, 2018).

Evidence of impacts of selective logging on HD species

The impacts of selective logging on HD species can be reliably anticipated to more severe the older the forest, as the biodiversity value of old-growth forests has been shown by many authors. A meta-analysis of 49 published studies on biodiversity in managed versus unmanaged forests in Europe (Paillet et al, 2010) found that bryophytes, lichens, fungi and saproxylic beetles were negatively affected by forest management. The study found that the global difference in species richness between managed and unmanaged forests increased with time since abandonment and indicated a gradual recovery of biodiversity. Amongst the various forest management pressures, the study found that the overall effect of selective cutting was significantly negative for bryophytes, lichens and saproxylic beetles (Paillet et al, 2010). (Joelsson et al, 2017)

There are indications that the impact of pressures from logging of old-growth forests, as well as removal of dead or dying trees (see below), on rare **bryophytes** is significant. A study of epiphytic bryophytes and lichens in beech forests in Sweden (Fritz and Brunet, 2010) found that at stand level, the availability of substrates, a high stand age and forest continuity were the most important factors explaining high species number of epiphytes of conservation concern (i.e. red-listed and indicator species). Within stands, plots containing old trees, at the base of slopes and with low recent forestry impact had the highest number of species. Additional evidence was provided by a study that compared diversity of epiphytic lichens and bryophytes (on trees, deadwood, soil and rocks) in two Mediterranean deciduous forest types (*Fagus sylvatica* and *Quercus cerris* forests) in old growth stands (containing individual trees over 50 cm in diameter) and selectively cut stands (Brunialti et al, 2010). The authors found higher species richness and a higher presence of rare lichen and bryophyte species in the old growth stands.

The Habitats Directive lists a number of **non-vascular and vascular plants** that are restricted to undisturbed old growth forest:

- *Cephalozia macounii* is a good indicator of old growth forests in the boreal region as it is restricted to very old (100 year or more) decaying wood from conifers in old-growth wet coniferous forests[85]. The critically endangered European population is considered to be severely fragmented since it is found in isolated and very small subpopulations, and sporophytes only occur extremely rarely, therefore the subpopulations may go extinct with a reduced possibility of recolonization, and there is a continuous decline in habitat quality and area.

- *Buxbaumia viridis* is a boreal-montane moss species which prefers primeval forests with coniferous wood on which it grows, requiring high humidity and fallen fir or spruce logs in advanced stages of decay. The species is restricted to forests under strict protection (i.e. with no management) (Voncina, Cykowska and Chachula, 2011) or forest remnants in inaccessible areas, where it is under threat from forest exploitation or other interventions (Köckinger, 2014). It is infrequent in managed forests because of its dependence on a humid environment under a closed canopy with high levels of dead wood in advanced stages of decay (Söderström, 1988).
- *Orthotrichum rogeri* appears to have contradictory ecological requirements, in that although it is a very rare moss, it is found both in pristine, old-growth and ancient forests with a natural dynamic of disturbance (wind throw, senescent fallen trees, etc.) where it colonizes young shrubs or young or upper branches of trees, or very small branches at the base of the trunk, and also in wooded montane pastures on the bark of mesic-mesohygric *Salix-Sambucus* trees in prominent edge conditions (enhanced wind velocity, amount of light and aggravated rate of evaporation) (Kiebacher et al, 2017; Poncet, Hugonnot and Vergne, 2015).
- The fern *Trichomanes speciosum* is restricted to damp closed canopy forests with very low levels of disturbance, and is highly sensitive to any type of forest management (Criado et al, 2017).

EU HD bark beetles are highly associated with old-growth forests. For instance, in a study in the Białowieża Forest (Poland), *Boros schneideri* showed preferences for old tree stands (over 140 years old) (Gutowski et al, 2014). The beetle *Rhysodes sulcatus* is known to be associated with the presence of large, moist and well rotten fallen logs with a diameter greater than 60 cm (Kostanjsek et al, 2018). All these species have also been reported by a number of Member States as under pressure from the removal of dead and dying trees, as shown in Figure 3-20. A study in Finland found that the HD beetle *Pytho kolwensis* is restricted to virgin spruce-mire forests with a stand continuity of at least 170–300 years, and a high volume of dead wood (73–111 m3/ha) (Siitonen and Saaristo, 2000). The authors suggest that *P. kolwensis* is mainly restricted to spruce-mire forests because of the long-term continuous availability of suitable host trees in these habitats. However, selective logging may have a more limited impact on the bark beetles *Cucujus cinnaberinus* and *Rosalia alpina* since these species prefer a certain degree of openness (Horak, Chumanova and Hilszczanki, 2012; Russo, Cistrone and Garonna, 2011a).

The list of threatened (i.e. red-listed) **terrestrial molluscs** contains a high proportion of species that are restricted to old growth forest and highly sensitive to logging (Cuttelod, Seddon and Neubert, 2011). A survey of native beech and oak forests in the Bavarian Forest National Park, Germany (Moning and Müller, 2009), found that the proportion of red-listed

[85] Konstantinova, N. 2019. Cephalozia macounii. The IUCN Red List of Threatened Species 2019: e.T87491664A87753094. Downloaded on 31 October 2019. ArtDatabanken, 2019. Artfakta. https://artfakta.se/artbestamning/taxon/cephalozia-macounii-293

mollusc species (compared to non-threatened species) remained the same along a gradient of increasing forest age from 50 up to around 350–400 years, but the number of species per plot increased significantly. Colonisation of disturbed forest patches by molluscs may occur considerably more rapidly if only small patches are affected and (micro) refugia remain close by. Donor sites would probably have to exceed the age range of 180–230 years (Moning and Müller, 2009).

Numerous **bat species** are known to depend on the presence of tree holes, particularly those available as a result of the activities of hole-creating woodpeckers, but also due to dampness that penetrates into the trunk through broken branches (Dietz, von Helversen and Nill, 2009). A study in Italy caught fewer Barbastelle bats (*Barbastellus barbastella*) in selectively logged forest than unmanaged forest, and fewer bats roosted in the logged forest (Russo et al, 2010). However, another study concluded that Barbastelles and other bat species are poor indicators of the effects of selective or restricted logging if the remaining forest area provides enough roosts, as some species benefit from the increased forest edge and open habitat areas, whilst the closed canopy species are unaffected (Mehr et al, 2012).

Evidence of impacts of selective logging on BD birds

The removal of large mature trees through logging is likely to have detrimental impacts on a wide range of forest BD birds, including such species as woodpeckers (Wesołowski and Martin, 2018). For example, the Black Woodpecker (*Dryocopus martius*) is found in a range of types of forest but they must have regular large mature forest stands (Garmendia, Cárcamo and Schwendtner, 2006). White-backed Woodpecker (*Dendrocopos leucotos*) and Three-toed Woodpecker (*Picoides tridactylus*) occupancy probability increased with lower logging intensity (on difficult-to-access slopes) and higher amount of dead and dying wood in Polish Carpathian forests (Kajtoch, Figarski and Pelka, 2013). A large number of raptors, and the Black Stork (*Ciconia nigra*) are also highly dependent on the presence of large trees for nest sites.

The endemic and vulnerable Corsican Nuthatch (*Sitta whiteheadi*) is in part threatened by forestry as the large trees suitable for the species are also favoured by the logging industry and since the 1970s local foresters have attempted to rejuvenate the pine forest by shortening the logging rotation, reducing the size of trees available for the species (Bourcet, 1996), cited in (Birdlife International, 2019b). In addition, it has been predicted that whenever an area larger than 2 ha is logged in a forest stand suitable for the species, a potential territory is likely to disappear for more than a century due to the slow growth of Corsican pine (Thibault et al, 2011).

3.4.3 Thinning of small and intermediate sized successional trees or removal of forest undergrowth

General biodiversity impacts

One of the commonest and first introduced management practices to improve tree growth and/or remove unwanted trees is thinning. It consists of the felling of a proportion of small and medium sized trees to allow the remaining trees to grow faster and become bigger as they receive more sunlight and space. The chosen trees for felling are those that are diseased or have low value (i.e. with less straight stems, with cracks or defects, tight and weak forks or the less vigorous and less healthy), but that can interfere with the growth of more valuable trees. There are several different methods, from systematic line thinning where entire rows are removed, to more time intensive systems that focus on the trees left within the canopy. Typically, between 15% and 30% of trees are removed[86].

Evidence of impacts on HD habitats

The Habitats Directive Annex I habitats are naturally regenerating natural or semi-natural forests, which do not require harvesting and associated management activities to maintain their biodiversity value. The only cases where they may commonly do so are if the trees have been harvested and/or planted at the same time and have an even age structure and/or need thinning. The maintenance of a diversity of age in tree population has been highlighted in the literature as a common requirement for the quality of forests including HD forest habitats (BfN, 2013).

Whilst fire is part of the natural ecology of boreal and Mediterranean natural forests, it can be detrimental to biodiversity under some current circumstances, e.g. where the forest habitats and species of EU interest occur in a few highly fragmented forest areas. Furthermore, climate change is altering the situation faster than natural forests can adapt or migrate. Consequently, Member States are reporting that Mediterranean forest habitats are under pressure from burning down, but at the same time some are reported as under pressure from removal of forest undergrowth (Figure 3-14) and dead and dying trees (Figure 3-20). The situation is therefore complex and it is beyond the scope of this study to draw conclusions on the potential interactions between bioenergy production and forest fire management and their impacts on habitats and species of EU interest.

Some forest types were created by traditional management systems that maintain an open forest structure, including coppicing, pollarding, and forest pastures or meadows[87]. These forest habitat types provide suitable habitat for invertebrates (such as butterflies) and plant communities that require an open canopy, and also host different bird communities to closed forest. If the management is abandoned, the forest will slowly change in structure and potentially in the long-term into a different forest type (European Commission, 2015a). Forest habitats of Community interest that have been established by historical and current management and which would disappear or change into other forest types under non-intervention management will require the continuation of active management (Müllerová, Hédl and Szabó, 2015).

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86 https://www.forestry.gov.uk/forestry/beeh-a8zfvj

87 Annex I habitats of this type include Fennoscandian wooded pastures (H9070), sub-Atlantic and medio-European oak or oak-hornbeam forests of the Carpinion-betuli (H9160), Galio-Carpinetum oak-hornbeam forests (H9170), Castanea sativa woods (H9260), Quercus suber forests (H9330)

Figure 3-14 Habitats Directive Annex I habitats reported as under high pressure from thinning of tree layer

Key to biogeographical regions: ALP Alpine; ATL Atlantic; BLS Black Sea; BOR Boreal; CON Continental; MAC Macaronesian; MED Mediterranean; PAN Pannonian; STE Steppic.
Evidence of sensitivity according to the literature – see text for related reference sources.

Habitat code	Habitat name	Member State	Evidence in the literature
9120	Atlantic acidophilous beech forests with Ilex and sometimes also Taxus in the shrublayer (Quercion robori-petraeae or Ilici-Fagenion)	ES (ATL)	
9180	Tilio-Acerion forests of slopes, screes and ravines	AT (ALP)	
91D0	Bog woodland	LV (BOR)	
91E0	Alluvial forests with Alnus glutinosa and Fraxinus excelsior (Alno-Padion, Alnion incanae, Salicion albae)	AT (CON)	
91G0	Pannonic woods with Quercus petraea and Carpinus betulus	AT (CON)	
91I0	Euro-Siberian steppic woods with Quercus spp.	AT (CON)	
9530	(Sub-) Mediterranean pine forests with endemic black pines	AT (ALP)	

Figure 3-15 Habitats Directive Annex I habitats reported as under high pressure from removal of forest undergrowth

Key to biogeographical regions: ALP Alpine; ATL Atlantic; BLS Black Sea; BOR Boreal; CON Continental; MAC Macaronesian; MED Mediterranean; PAN Pannonian; STE Steppic.
Evidence of sensitivity according to the literature – see text for related reference sources.

Habitat code	Habitat name	Member State	Evidence in the literature
9120	Atlantic acidophilous beech forests with Ilex and sometimes also Taxus in the shrublayer (Quercion robori-petraeae or Ilici-Fagenion)	ES (ATL)	
9160	Sub-Atlantic and medio-European oak or oak-hornbeam forests of the Carpinion betuli	IT (CON)	
91G0	Pannonic woods with Quercus petraea and Carpinus betulus	DE (CON)	

Evidence of impacts on HD species

Three Member States reported a high pressure of thinning of the tree layer on HD species including bryophytes and the fern *Trichomanes speciosum*, bat and beetle species and a mollusc (Figure 3-16). 13 Member States reported high pressure of removal of forest undergrowth on 35 HD species or taxa.

Thinning forests may increase the extinction rate of specialist epixylic **bryophytes** that depend on closed canopy conditions (as mentioned above for selective logging), but also have the effect of increasing the overall epiphytic species richness due to the creation of patches of dry exposed conditions. A study in oak-rich temperate forest in southern Sweden found that the pooled frequency of lichen and bryophyte species of conservation concern on large oaks increased at the 1 ha plot scale nine years after the thinning, although the change in species composition was weak (Nordén et al, 2012). The study concludes that conservation-oriented partial cutting in temperate mixed forest seems favourable for the epiphytic lichen and bryophyte flora on large trees. A previous study in the same forest type found that thinning increased the lichen species density on forest stumps (epixylic) but also increased the extinction rate of some epixylic bryophyte species due to the switch to dry exposed conditions (Paltto, Nordén and Götmark, 2008).

Removal of forest undergrowth is reported as a high pressure on a number of **HD bats**. For example, *Myotis bechsteinii*, a widespread species in the temperate beech forest zone of west, central and east Europe, occurs preferentially in structure-rich forests with a pronounced species-rich shrub layer (Dietz, von Helversen and Nill, 2009). They hunt their prey mostly at low levels close to vegetation, with a high proprortion of insects gleaned directly from the vegetation.

The HD **dormouse** *Muscardinus avellanarius* prefers mid-height woodland habitat (5–10m tall), with low proportions of high forest (over 10m tall), for both ranging and resting sites, often along woodland edges and in relatively dense vegetation. However, they also require the presence of some mature trees with tree cavities, squirrel or bird nests to make their own nests in. A study in the UK indicates that dormice require the mid-successional and forest edge habitats created by felling, but suffer from the removal of forest undergrowth, for example for tree replanting (Goodwin et al, 2018). Similarly, the forest dormouse *Dryomys nitedula* prefers nest sites with a better developed and diverse understorey (especially with young rowan, lime and aspen trees), with more abundant mature oak, lime and black alder trees and a higher percentage of raspberry and bramble cover, as well as overgrown clearings, as shown in a study in Lithuania (Juškaitis, Balciauskas and Augutė, 2012).

In contrast, managing woodlands for most of the **HD Lepidoptera** requires the maintenance of relatively low tree density and/or permanent or dynamically managed clearings, in a diverse structure including some mature or tall trees, some dense regrowth, numerous sunny rides and glades (both large and small) and patches of recently cleared and regenerating open areas with sparse ground vegetation and warm unshaded conditions. Although two Member States reported removal of undergrowth as a high pressure on two HD Lepidoptera, European woodland Lepidoptera species as a rule are reliant on open sunny habitats within forests and woodland, such as sparse stands, bogs, streamsides, clearings, rides or edges (Settele et al, 2009), and are likely to benefit from the creation of open areas. This includes HD species such as Scarce Fritillary (*Euphydrias maturna*), which depends on clearings with wet grassland within woodland, and Freyer's Purple Emperor (*Apatura metis)*, which relies on willow (*Salix alba*) along wooded riverbanks in warm damp places. A study of the habitat requirements of *Euphyrias maturna* populations in Austria, the Czech Republic and Germany (Hjältén, Stenbacka and Andersson, 2010) found that larvae of Central European populations develop on saplings and low-hanging branches of Ash (*Fraxinus* spp.) located in sunny but damp conditions, and concludes that the restoration of coppicing and forest pasture methods offers the only chance for its survival in Central Europe. Woodland Brown (*Lopinga achine*) are typical coppice species as they prefer forests with a light canopy on south-facing slopes (Streitberger et al, 2012), with a delicate balance between sunlight and humidity. Larvae develop on *Carex montana* growing under 60-80% canopy cover, as traditionally maintained by cattle grazing in woodland (refs cited in (Settele et al, 2009). A LIFE project that opened and enlarged clearings in mixed forests in Belgium for Marsh Fritillary (*Euphydrias aurinia*) resulted in the colonisation of clearings by new populations, thereby expanding the meta-population and compensating for the decline of existing populations (European Commission, 2012).

Saga pedo was reported by Spain as under pressure from thinning. It is a bush-cricket associated with patches of dense herbaceous vegetation on grasslands, and can be affected by the large-scale removal of scrub and small trees, although overall, the impact of succession on unmanaged grasslands is much greater and the species is therefore dependent on the removal of trees from overgrown sites (Krištín and Kaňuch, 2007).

Figure 3-16 Habitats Directive Annex II, IV and V species reported as under high pressure from thinning of tree layer

Key to biogeographical regions: ALP Alpine ATL Atlantic BLS Black Sea BOR Boreal CON Continental MAC Macaronesian MED Mediterranean PAN Pannonian STE Steppic
Evidence of sensitivity according to the literature – see text for related reference sources.

Taxonomic group	Species name	Member State	Evidence in the literature
Mammal (Bat)	*Myotis alcathoe*	SE (CON)	
Beetle (bark)	*Cucujus cinnaberinus*	PL (ALP,CON)	
Beetle (bark)	*Rosalia alpina*	ES (ALP,ATL,MED)	
Beetle (ground)	*Rhysodes sulcatus*	PL (ALP,CON)	
Invertebrate	*Saga pedo*	ES (MED)	
Mollusc	*Geomalacus maculosus*	ES (ATL,MED)	
Plant (bryophyte)	*Buxbaumia viridis*	ES (ALP,MED)	SE
Plant (bryophyte)	*Dicranum viride*	PL (CON,ALP)	SE
Plant (vascular)	*Trichomanes speciosum*	PL (CON)	

Figure 3-17 Habitats Directive Annex II, IV and V species reported as under high pressure from removal of forest undergrowth

Key to biogeographical regions: ALP Alpine ATL Atlantic BLS Black Sea BOR Boreal CON Continental MAC Macaronesian MED Mediterranean PAN Pannonian STE Steppic
Evidence of sensitivity according to the literature – see text for related reference sources.

Taxonomic group	Species name	Member State	Evidence in the literature
Mammal (bat)	*Barbastella barbastellus*	BG (ALP, CON, BLS)	
Mammal (bat)	*Myotis bechsteinii*	BE (ATL, CON) BG (CON, ALP) DE (ATL) LU (CON)	
Mammal (bat)	*Myotis brandtii*	DE (ATL) LU (CON)	
Mammal (bat)	*Myotis emarginatus*	LU (CON)	
Mammal (bat)	*Myotis mystacinus*	DE (ATL)	
Mammal (bat)	*Pipistrellus nathusii*	DE (ATL)	
Mammal (bat)	*Pipistrellus pipistrellus*	DE (ATL)	
Mammal (bat)	*Pipistrellus pygmaeus*	DE (ATL)	
Mammal (bat)	*Plecotus austriacus*	DE (ATL)	
Mammal (bat)	*Vespertilio murinus*	DE (ATL)	
Mammal	*Castor fiber*	DE (ATL)	
Mammal	*Dryomys nitedula*	LT (BOR), PL (ALP, CON)	
Mammal	*Felis silvestris*	PT (MED)	
Mammal	*Muscardinus avellanarius*	BE (ATL, CON) DE (ATL, CON) DK (CON) HU (PAN) IT (MED) NL (ATL) PL (ALP, CON)	
Mammal	*Sicista betulina*	PL (CON, ALP)	
Arthropod (butterfly)	*Euphydryas aurinia*	ES (ALP, ATL, MED)	
Arthropod (moth)	*Proserpinus proserpina*	HU (PAN)	

Arthropod	*Gomphus graslinii*	ES (ATL, MED)	
Arthropod	*Lucanus cervus*	ES (ALP, ATL, MED)	
Arthropod	*Rhysodes sulcatus*	PL (CON, ALP)	
Invertebrate (other)	*Geomalacus maculosus*	ES (ATL, MED)	
Plant (vascular)	*Asplenium hemionitis*	PT (MED, MAC)	
Plant (vascular)	*Centaurea pinnata*	ES (MED)	
Plant (vascular)	*Doronicum plantagineum ssp. tournefortii*	PT (MED)	
Plant (vascular)	*Euphorbia transtagana*	PT (MED)	
Plant (vascular)	*Galanthus nivalis*	IT (MED, CON)	
Plant (vascular)	*Narcissus asturiensis*	ES (ALP, MED)	
Plant (vascular)	*Petagnia saniculifolia*	IT (MED)	
Plant (vascular)	*Pyrus magyarica*	HU (PAN)	
Plant (vascular)	*Ruscus aculeatus*	ES (ALP, ATL, MED)	
Plant (bryophyte)	*Buxbaumia viridis*	ES (ALP, MED)	
Plant (bryophyte)	*Dicranum viride*	HU (PAN)	
Plant (bryophyte)	*Leucobryum glaucum*	LV (BOR)	
Plant (bryophyte)	*Orthotrichum rogeri*	ES (ALP)	
Plant (lichen)	*Cladonia spp. (subgenus Cladina)*	LV (BOR)	

Evidence of impacts on BD birds

The published literature indicates that bird species differ considerably in their response to thinning, and several studies have found no measurable effects (Fuller, 2013). However, evidence from the Birds Directive reporting (Figure 3-18), suggests that a number of species are detrimentally affected by thinning. Griesser et al (2007) also found that in Fennoscandian forests thinning can have a negative impact on some bird species by removing much of the understorey and, thus reducing habitat quality. It is also possible that thinning of semi-natural forests in Norway led to an increase in predation of Black Grouse by Northern Goshawk (*Accipter gentilis*) (Wegge and Rolstad, 2011). In the Mediterranean region, the Annex I species Olive-tree Warbler *(Hippolais olivetorum)* is considered to be threatened by changes in habitat structure, especially from clearance and thinning of woodland (Birdlife International, 2015).

In some circumstances, such as where planted forests have been left unmanaged and develop dense and even aged structures, thinning may be beneficial for some species. Also it has been shown that thinning of conifer plantations can be beneficial for the Western Capercailie (*Tetrao urogallus*), as the increase in light increases the abundance of the shrub *Vaccinium myrtillus*, which is an important food source (Broome, Connolly and Quine, 2014). However, this may be dependent on thinning levels that are greater than under normal commercial practices.

Figure 3-18 Birds Directive Annex I species reported as under high pressure from thinning of the tree layer

Evidence of sensitivity according to the literature – see text for related reference sources.

Species name	**Member State**	**Evidence in the literature**
Accipiter gentilis arrigonii	IT	
Aegolius funereus	SI	
Aquila clanga	FI	
Ciconia nigra	PT, RO	
Circaetus gallicus	PT, RO	

Dendrocopos leucotos	SI	
Hieraaetus pennatus	PT	
Hippolais olivetorum		Yes - detrimental
Milvus milvus	PT	
Picoides tridactylus	SI	
Lyrurus / Tetrao tetrix		Norway - detrimental
Tetrao urogallus		Yes – beneficial

3.4.4 Removal of dead and dying trees and dead wood

General biodiversity impacts

Another common forest management practice in more intensively managed forests is the removal of dead or dying trees, for example in an attempt to reduce pest and disease outbreaks and, in Mediterranean forests, to reduce the risks of fires and their severity. As a result, the current volumes of deadwood in managed forests in Europe are normally less than 10% of natural levels (Stokland, Siitonen and Jonsson, 2012). This has a substantial effect on the forest ecosystem and its associated plant and animal communities. The quantity and quality (structure, microclimate, diversity) of dead wood is a critical factor for many species of conservation concern in European forests, including saproxylic bryophytes, lichens, fungi and invertebrates. There is a strong body of scientific evidence showing the importance of deadwood and a range of related factors for many species, and evidence that deadwood volume and diversity in most managed forests is currently too low to maintain species richness and conserve HD species (Hahn and Christensen, 2004; Merganicova et al, 2012).

Large-diameter deadwood is currently the most limiting factor in managed forests, and it is critical for a number of species of conservation concern, but there are also rare saproxylic species significantly associated with small diameter deadwood, which could become limiting if the exploitation of harvest residues is intensified. As different species specialise in different environmental conditions, it is important that the heterogeneity of deadwood substrates is retained, in both sunny and shady forest conditions.

A review of dead-wood threshold data from a range of European forest types revealed 36 critical values for minimum amounts of deadwood to support biodiversity, with ranges of 10–80 m^3 ha^{-1} for boreal and lowland forests and 10–150 m^3 ha^{-1} for mixed-montane forests, with peak values at 20–30 m^3 ha^{-1} for boreal coniferous forests, 30–40 m^3 ha^{-1} for mixed-montane forests, and 30–50 m^3 ha^{-1} for lowland oak–beech forests (Mueller and Bütler, 2010). Relying on the passive self-restoration of deadwood that follows the abandonment of forest activities would take a long time to restore amounts of deadwood to minimum thresholds, so many HD forest habitats that have been intensively managed require active measures to increase deadwood levels (Bouget et al, 2014).

To mitigate the declines in deadwood volumes and associated biodiversity resulting from forestry operations, especially clearcutting, it has become required practice in some countries to retain a proportion of live and dead trees at the time of harvest. The aim of this is to maintain some level of continuity in forest structure and composition, as well as providing a refuge for some deadwood species that may then be able to recolonise surrounding forest areas as they mature. A study of the effects of this retention practice in Sweden, where it has been carried out for over 25 years as a result of legal requirements and certification standards, found that it had increased deadwood volumes compared to previous practices (Kruys et al, 2013). However, the estimated current average level was only 8 $m^3 ha^{-1}$, which is less than the critical threshold for boreal forests (as noted above). Furthermore, whilst another study found that the volume of deadwood had increased since the mid-1990s (by 25% to 7.6 $m^3 ha^{-1}$), the level of heterogeneity of deadwood types was low in terms of species, diameter and decay classes (Jonsson et al, 2016). Furthermore, the study considered that the increase in deadwood was not primarily the result of the retention forestry policy, but instead a result of storm events.

Evidence of impacts from removal of dead and dying trees on HD habitats

Many of the HD forest types are characterised by high levels of veteran and dead trees and dead wood, habitats for host-specific saproxylic species and their specialist parasitoids and predators (European Commission, 2013). Therefore, the removal of dead and dying trees can have a devastating effect on HD habitat types since it can affect fundamental characteristics of the habitat and its ecological processes.

Most beech forest types in Europe are particularly lacking in deadwood structures as they have been managed for the last two centuries in shelterwood systems targeting wood production, in which the canopy is substantially thinned to enable dense natural regeneration, after which the remaining standing trees are cut. This system results in large, single-layered and even-aged beech stands in 90 to 140 year rotations. A review of comparative studies of managed and unmanaged beech forests in Central Europe (Brunet, Fritz and Richnau, 2010) found that shelterwood managed forests generally lack their typical epiphytic lichen and bryophyte community that develops on trees over 180 years old, epixylic bryophytes on dead wood, and many saproxylic fungi, although they retain their spring geophyte and shade-tolerant summer vascular plants.

Presumably, the effects of shelter wood systems are similar in other forest types.

Member State reporting data clearly show that many habitat types are being impacted by reductions in deadwood and dying trees in forests (**Error! Reference source not found.** 4-15). A ustria, Belgium, and Germany frequently reported HD forest habitats impacted by the removal of dead and dying trees. In Germany an insufficiency of deadwood is one of the key factors constraining the achievement of favourable conservation status in forest habitats, as well as their limited diversity of native tree species, tree ages and lack of open spaces (BfN, 2013), although the situation has improved over the last decade, as shown by the third forest inventory (BMEL, 2013).

Figure 3-19 Habitats Directive Annex I habitats reported as under high pressure from removal of dead and dying trees

Key to biogeographical regions: ALP Alpine; ATL Atlantic; BLS Black Sea; BOR Boreal; CON Continental; MAC Macaronesian; MED Mediterranean; PAN Pannonian; STE Steppic.
Evidence of sensitivity according to the literature – see text for related reference sources.

Habitat code	Habitat name	MS reporting high pressure	Evidence in the literature
2180	Wooded dunes of the Atlantic, Continental and Boreal region	LV, SE	
9010	Western Taïga	LV	
9110	Luzulo-Fagetum beech forests	AT, BE, NL	Central Europe
9120	Atlantic acidophilous beech forests with Ilex and sometimes also Taxus in the shrublayer (Quercion robori-petraeae or Ilici-Fagenion)	BE, DE, ES, NL	
9130	Asperulo-Fagetum beech forests	AT, BE	Central Europe
9140	Medio-European subalpine beech woods with Acer and Rumex arifolius	AT	
9150	Medio-European limestone beech forests of the Cephalanthero-Fagion	AT, BE, DE	Central Europe
9160	Sub-Atlantic and medio-European oak or oak-hornbeam forests of the Carpinion betuli	AT, BE, DE	
9170	Galio-Carpinetum oak-hornbeam forests	AT	
9180	Tilio-Acerion forests of slopes, screes and ravines	AT, BE	
9190	Old acidophilous oak woods with Quercus robur on sandy plains	BE, DE	
91E0	Alluvial forests with Alnus glutinosa and Fraxinus excelsior (Alno-Padion, Alnion incanae, Salicion albae)	AT, BE, DE, NL	
91F0	Riparian mixed forests of Quercus robur, Ulmus laevis and Ulmus minor, Fraxinus excelsior or Fraxinus angustifolia, along the great rivers (Ulmenion minoris)	AT, BE, DE	
91G0	Pannonic woods with Quercus petraea and Carpinus betulus	AT, DE	
91K0	Illyrian Fagus sylvatica forests (Aremonio-Fagion)	AT	
91U0	Sarmatic steppe pine forest	DE	
9530	(Sub-) Mediterranean pine forests with endemic black pines	AT	
9560	Endemic forests with Juniperus spp.	ES	

Evidence of impacts from removal of dead and dying trees on HD species

Numerous countries have reported a high pressure from removal of dead and dying trees on HD species, particularly bats and beetles (Figure 3-19). A comparison of **epixylic bryophytes** in managed coniferous, managed deciduous and unmanaged deciduous forests found that species richness of epixylic bryophytes was strongly limited by the considerable lack of deadwood in all forest types, even in currently unmanaged forests (Bergmeier, Petermann and Schröder, 2010). For example, the moss *Buxbaumia viridis* was reported by several countries as being potentially highly impacted by the removal of dead and dying trees. A survey of lichen and bryophyte communities in the UK found that lichen species-richness was lower in mature non-native Sitka Spruce (*Picea sitchensis*) and native pine (*Pinus sylvestris*) plantations compared to semi-natural pine and oak (*Quercus robur*) woodlands; in particular the frequency of *Cladonia* lichen species was associated with the frequency of decorticate snags (particularly in pine plantations) and stumps (Humphrey et al, 2002). Bryophyte species-richness was positively correlated with large diameter (>20 cm), well-decayed logs and stumps (Humphrey et al, 2002).

- *Plagiomnium drummondii* is assessed as endangered in Europe, primarily due to removal of dead wood and and harvesting in mature spruce and aspen forests[88].

Within the group of saproxylic beetles, each species has a different critical threshold of deadwood availability and structure below which the species becomes extinct, and species richness increases more or less continually with increasing deadwood habitat (Bouget et al, 2014; Brin et al, 2011; Horák, Vávrová and Chobot, 2010; Jonsell, Hansson and Wedmo, 2007; Lassauce, Lieutier and Bouget, 2012).

Large-diameter deadwood is currently particularly lacking in intensively managed forests and is therefore associated with most endangered **beetle species**. *Agathidium pulchellum* feeds exclusively on a slime-mould species associated with mid-decayed aspen, spruce and birch logs in boreal forests, but even if its host is present, the beetle is absent from sites with less than 80 aspen and spruce logs per hectare (Laaksonen et al, 2010). *Cerambyx cerdo* requires large pieces and is also an important habitat engineer for other species because it produces large galleries (Brin et al, 2011). Authors recommend increasing deadwood to over 20 m^3 ha-1 and increasing the proportion of large (over 50cm) diameter logs (Moning and Müller, 2009). A Swedish study found other rare species were significantly associated with small diameter deadwood (branches) (Jonsell, Hansson and Wedmo, 2007). A meta-analysis indicated that deadwood volume and saproxylic beetle species richness are more correlated in boreal forests than in temperate forests, but this may be due to a lack of evidence from temperate forests (Lassauce et al, 2011). Old oak trees are of particular importance for saproxylic species in Mediterranean oak forests because they provide special habitat features such as hollows and various kinds of fungi-infested wood (Remm and Lõhmus, 2011). Reinstating pollarding of trees can promote the formation of tree hollows in certain tree species, making the trees attractive for saproxylic biodiversity (Sebek et al, 2013).

The importance of deadwood has been revealed for the threatened amphibian the Golden Alpine Salamander (*Salamandra atra aurorae*), which is endemic to a small portion of the Italian Alps (Romano et al, 2018).

Central European beech forests (habitat type 9130 Asperulo-Fagetum) are currently poor habitats for **forest-dwelling bats**, particularly for the HD species that roost in tree cracks and snags (*Barbastella barbastellus*, *Myotis bechsteinii*, *Nyctalus noctula, Nyctalus leisleri, Pipistrellus nathusii)* (Zehetmair et al, 2015). Due to a long history of timber extraction under the shelterwood system, these forests are characterised by a low abundance of veteran trees with cracks, holes, and snags. A German study found no significant difference in the quality of the habitat for bats between beech forests within and outside Natura 2000 sites, indicating that the forest management within Natura 2000 has not yet significantly changed conditions in favour of bats (Zehetmair et al, 2015).

Tree hollows are key breeding habitats for a number of **other HD mammals**, including the Siberian Flying Squirrel (*Pteromys volans*). A study has shown that there is a dramatic reduction in occupancy after clear-felling even when the actual tree with the hole is not

[88] Baisheva, E., Ignatov, M., Konstantinova, N., Maslovsky, O., Sabovljevic, M. & Ştefănuţ, S. 2019. Plagiomnium drummondii. The IUCN Red List of Threatened Species 2019: e.T84318278A87756338. Downloaded on 31 October 2019.

felled, compared to similar unfelled forest (Santangeli et al, 2013). . Deadwood piles, tree stumps and tree hollows are key refuges and resting places for forest carnivores of conservation concern such as Wild Cat (*Felix sylvestris*) and Genet (*Genetta genetta*) in Mediterranean *Quercus ilex* and *Quercus suber* forests (Carvalho et al, 2014). The authors of the Genet study recommend that forest management should maintain large riparian trees and safeguard at least 4.6 hollow-bearing trees per 100 ha.

Figure 3-20 HD species reported as under high pressure from removal of dead and dying trees and evidence in the literature

Key to biogeographical regions: ALP Alpine; ATL Atlantic; BLS Black Sea; BOR Boreal; CON Continental; MAC Macaronesian; MED Mediterranean; PAN Pannonian; STE Steppic.
Evidence of sensitivity according to the literature – see text for related reference sources.

Taxonomical group	Species name	MS reporting high pressure	Evidence in the literature
Mammal (Bat)	*Barbastella barbastellus*	BE, BG, CZ, ES, HU, IT, LU, PL, RO	EU
Mammal (Bat)	*Myotis alcathoe*	CZ, DE, ES, HU, SE	EU
Mammal (Bat)	*Myotis bechsteinii*	BE, CZ, DE, ES, HU, IT, LU, PL, RO	EU
Mammal (Bat)	*Myotis brandtii*	AT, BE, CZ, EE, HU, LU	EU
Mammal (Bat)	*Myotis daubentonii*	BE, CZ, EE, PL, RO,	EU
Mammal (Bat)	*Myotis mystacinus*	BE, CZ, EE, ES, HU, RO	EU
Mammal (Bat)	*Myotis nattereri*	BE, CZ, EE, HU, PL, RO	EU
Mammal (Bat)	*Nyctalus lasiopterus*	CZ, ES, HU, IT	EU
Mammal (Bat)	*Nyctalus leisleri*	AT, BE, CZ, DE, ES, HU, IT, LU, PL	EU
Mammal (Bat)	*Nyctalus noctula*	BE, CZ, EE, ES, HU, IT, LT, LU, PL, RO	EU
Mammal (Bat)	*Pipistrellus pygmaeus*	CZ, EE, ES, HU	EU
Mammal (Bat)	*Plecotus auritus*	BE, CZ, DE, EE, ES, HU, IT, RO	EU
Mammal	*Pteromys volans*		FI
Mammal	*Felix sylvestris*		PT
Mammal	*Genetta genetta*		PT
Amphibian	*Salamandra atra aurorae*		IT
Beetle (bark)	*Boros schneideri*	EE, LT, PL, SK	PL
Beetle (bark)	*Cucujus cinnaberinus*	AT, CZ, DE, ES, HU, IT, LT, LV, PL, SI, SK	CZ
Beetle (bark)	*Rosalia alpina*	AT, CZ, DE, ES, HU, IT, SI, SK	IT
Beetle (ground)	*Rhysodes sulcatus*	AT, CZ, HU, IT, PL, SI, SK,	CZ, SI, HR
Beetle (saproxyllic)	*Cerambyx cerdo*	BE, CZ, DE, ES, HU, IT, PL, PT, SI, SK	EU
Beetle (saproxyllic)	*Limoniscus violaceus*	AT, CZ, DE, ES, HU, SK	EU
Beetle (saproxyllic)	*Lucanus cervus*	AT, BE, CZ, DE, ES, HU, IT, NL, PT, SI, SK	EU
Beetle (saproxyllic)	*Osmoderma eremita*	DE, EE, ES, HU, LT, PL, SI, SK,	EU
Plant (bryophyte)	*Buxbaumia viridis*	AT, BG, CZ, DE, ES, IT, SE, SI	EU

Evidence of impacts from removal of dead and dying trees on BD Birds

As indicated in Figure 3-21, a significant proportion of BD birds that are specialist forest species are under pressure from the removal of dead and dying trees. Not surprisingly these include species that are reliant on large dead trees for nesting on (e.g. Osprey) or cavities in deadwood for nesting in (e.g. some owls and many forest passerines). Deadwood is particularly important for hole-nesting species in broadleaved forests, as the birds depend more on decay rather than excavators to provide holes (Wesołowski and Martin, 2018). In addition, several of the species also depend on deadwood invertebrates as food resources, especially Black Woodpecker (*Dryocopus martius*) and the *Dendrocopus* woodpecker species. Tucker and Heath (1994) notes that the Three-toed Woodpecker *(Picoides tridactylus)* is threatened by large-scale commercial logging and modern forestry management practices including the removal of dead or insect-infested trees. Winkler and Christie (2002) found that intense forest management, leading to a reduction of dead wood or the introduction of conifers resulted in a reduced White-backed Woodpecker *(Dendrocopos leucotos)* population.

In some cases, the placement of artificial nest boxes can mitigate the loss of nesting sites for some species (Wesołowski and Martin, 2018). For example, the modern forestry practice of removing old trees and chimney-like stumps caused a decrease in the population of the Ural Owl (*Strix uralensis*), but provision of thousands of nest boxes has led to recovery and even an increase (Holt et al, 2019).

Figure 3-21 Birds Directive Annex I species reported as under high pressure from removal of dead and dying trees

Evidence of sensitivity according to the literature – see text for related reference sources.

BD species	MS reporting removal of dead and dying trees pressure	Evidence in the literature
Aegolius funereus	AT, BG, LT, RO, SI	
Certhia brachydactyla dorotheae	CY	
Coracias garrulus	AT, LT, SK	
Dendrocopos leucotos	AT, BG, ES, CZ, HU, LT RO, SI, SK	Yes
Dendrocopos medius	AT, BE, BG, CZ, IT, LT, LU, RO, SI	Yes
Dendrocopos syriacus	BG, SK	Yes
Dryocopus martius	AT, BG, IT, LT, LU, RO, SI	Yes
Ficedula albicollis	AT, SI	
Ficedula parva	AT, BG, SK	
Ficedula semitorquata	BG	
Glaucidium passerinum	BG, RO, LT, SI	
Pandion haliaetus	LT	
Picoides tridactylus	AT, BG, CZ, IT, LV, RO, SI	Yes
Picus canus	AT, BG, IT, LT, LU, SK	
Strix uralensis	BG, LT	Yes
Surnia ulula	SE	Yes

3.4.5 Effects of clear cutting

General biodiversity impacts

Clear cutting consists of the process of harvesting in which all trees are cleared from the site (or as noted above, a few may be retained to mitigate environmental impacts). Production begins with the logging of roundwood, the skidding of logs to a road and bark stripping. This process involves the disturbance of soil because of the physical operations. Stump removal is dealt with in section 3.4.6.

The clear cutting of forest areas alters tree species composition and the general structure of the forest and can also lead to nutrient depletion.

Evidence of impacts of clear cutting on HD habitats

Article 17 reporting shows that many habitat types aare under pressure from forest clearance (Figure 3-22). There is evidence of recent loss of old-growth Annex I forest habitats to clearcutting in Sweden (Sahlin, 2010), Mediterranean countries (Zaghi, 2008), and Natura 2000 sites in Romania (Knorn et al, 2013).

The case of Latvia is especially striking as it reported a high number of HD habitats as under high pressure from forest clearance over the 2007-2012 reporting period. The felling rate in state-owned forests increased by about 60% between 2009 and 2010, compared to that in the previous 3 years, due to an increase in the allowed felling volume of 4 million m^3 (Rendenieks, Nikodemus and Brūmelis, 2015). Much of this felled biomass was used for pellet production, and Latvia has been identified as the leading pellet exporter in the EU with a total production of 1,277 thousand tonnes in 2012 (Zwolinski, 2015). Latvia has also reported some non-forest habitats as under high pressure from forest clearance, notably lowland rivers and wetlands (springfens and petrifying springs).

Figure 3-22 Habitats Directive Annex I habitats reported as under high pressure from forest clearance

Key to biogeographical regions: ALP Alpine; ATL Atlantic; BLS Black Sea; BOR Boreal; CON Continental; MAC Macaronesian; MED Mediterranean; PAN Pannonian; STE Steppic.

Habitat code	Habitat name	Member State	Evidence in the literature
2180	Wooded dunes of the Atlantic, Continental and Boreal region	LV	
3260	Water courses of plain to montane levels with the Ranunculion fluitantis and Callitricho-Batrachion vegetation	LV	
7160	Fennoscandian mineral-rich springs and springfens	LV	
7220	Petrifying springs with tufa formation (Cratoneurion)	LV	
8310	Caves not open to the public	HU	
9010	Western Taïga	LV	
9020	Fennoscandian hemiboreal natural old broad-leaved deciduous forests (Quercus, Tilia, Acer, Fraxinus or Ulmus) rich in epiphytes	LV	
9050	Fennoscandian herb-rich forests with Picea abies	EE	
9110	Luzulo-Fagetum beech forests	AT, CZ, HU	
9120	Atlantic acidophilous beech forests with Ilex and sometimes also Taxus in the shrublayer (Quercion robori-petraeae or Ilici-Fagenion)	ES	
9130	Asperulo-Fagetum beech forests	CZ, HU	
9180	Tilio-Acerion forests of slopes, screes and ravines	LV	
91D0	Bog woodland	LV	
91E0	Alluvial forests with Alnus glutinosa and Fraxinus excelsior (Alno-Padion, Alnion incanae, Salicion albae)	BE, IT, LV	
91F0	Riparian mixed forests of Quercus robur, Ulmus laevis and Ulmus minor, Fraxinus excelsior or Fraxinus angustifolia, along the great rivers (Ulmenion minoris)	BG, CZ	
91G0	Pannonic woods with Quercus petraea and Carpinus betulus	DE	
91K0	Illyrian Fagus sylvatica forests (Aremonio-Fagion)	HU	
91M0	Pannonian-Balkanic turkey oak-sessile oak forests	HU	
9320	Olea and Ceratonia forests	PT	
9410	Acidophilous Picea forests of the montane to alpine levels (Vaccinio-Piceetea)	AT	
9430	Subalpine and montane Pinus uncinata forests (* if on gypsum or limestone)	ES	
9530	(Sub-) Mediterranean pine forests with endemic black pines		Yes

Evidence of impacts of clear-cutting on HD species

Most of the species and habitats protected by the EU Nature Directives are highly sensitive to clear-cutting. Some species groups, notably bats, butterflies, and certain bird species, can temporarily profit from areas opened up in forests. However, most of the species that benefit from clear-cutting are more generalist species that are not protected under the Nature directives. Appropriate timing and seasonality of silvi-cultural interventions can change the severity of impacts on bird and mammal species if done outside the breeding season, but is not likely to significantly change the severity of impacts on forest habitats, invertebrates, non-vascular or vascular plants of EU interest, or any vertebrate species that rely on tree micro-habitat refuges all year round (e.g. tree holes and splits, dead wood).

A study of clear-cutting of boreal stream-side forests (Dynesius and Hylander, 2007) showed a strong reduction in **liverwort** species richness and **bryophyte** species associated with convex substrates, especially woody debris species, and most of the negative effects of clear-cutting on bryophyte species persisted almost halfway into the next forestry rotation period. Furthermore, a follow up study found that after 15 years cut stands had significantly fewer species of liverworts, especially of species specialised to growing on wood or bark (Dynesius, 2015). A list of affected HD species is provided in Figure 3-23.

Forest clearance can significantly affect invertebrates in forests. For example, bark beetles such as *Cucujus cinnaberinus* and *Rosalia alpina* are highly associated with the outermost layers of stems and roots of woody plants and unmanaged felling activities can eliminate the habitat required for these species (Horák, Vávrová and Chobot, 2010; Russo, Cistrone and Garonna, 2011a).

The arboreal specialist **HD mammals** Siberian Flying Squirrel (*Pteromys volans*) and Hazel Dormouse (*Muscardinus avellanarius*) are negatively affected by clear-cutting in temperate and boreal forests in Europe as shown by a meta-analysis (Bogdziewicz and Zwolak, 2014). The squirrel prefers mature spruce-dominated (*Picea abies*) mixed forests, which often have high amounts of dead wood (Hurme et al, 2008), and clear-cut areas within 150 m of breeding sites and resting places substantially increase the risk of the species abandoning the habitat (Jokinen, Mäkeläinen and Ovaskainen, 2015). The recent population decline has been more rapid than the loss of habitat area (Koskimäki et al, 2014), indicating that disturbance is a key factor (Freese et al, 2006). The current retention guidelines protecting areas of 10-30m radius around the tree hole are not sufficient protection – a study concludes that the total effectiveness of this conservation measure is only c. 1.5% (Santangeli et al, 2013).

Bats are also known to be impacted by clear-cutting. Russo et al (2010) showed that at least small numbers of dead trees should be retained in logged areas to favour population expansion and landscape connectivity to conserve *Barbastella barbastellus*, which is in line with the findings obtained by other authors (eg (Müller et al, 2013) and the ecological requirements of many bat species such as *Myotis bechsteinii, Myotis nattereri, Nyctalus leisleri, Nyctalus noctula and Plecotus auritus* (Dietz, von Helversen and Nill, 2009). Bat occurrence and activity is significantly higher in forests which have a high canopy surface roughness and a range of canopy heights, because of the diversity of tree ages/sizes and tree species (Jung et al, 2012).

The Swedish Prioritised Action Framework[89], which identifies priorities for implementing Natura 2000, notes that the Mountain Hare (*Lepus timidus*) has been pushed back by the snow-free winters and clear cutting practices, which lead to increased predation and competition with the introduced European hare, whose range is expanding. Intensive forest management that is destroying bear habitat is having a negative impact on Brown Bear (*Ursus arctos*) in the Bulgarian and East Balkan population and the Pyrenees, and possibly in Scandinavia (Boitani et al, 2015).

Exclusion zones need to be established around carnivore dens to prevent disturbance, and ideally forest operations should be avoided in the vicinity or carried out between August and January when HD carnivores are not breeding (Boitani et al, 2015). This requires the mapping of key breeding sites, and the identification of the most critical feeding areas in spring, summer and autumn is also important (Boitani et al, 2015).

Figure 3-23 Habitats Directive Annex II, IV and V species reported as under high pressure from forest clearance

Key to biogeographical regions: ALP Alpine; ATL Atlantic; BLS Black Sea; BOR Boreal; CON Continental; MAC Macaronesian; MED Mediterranean; PAN Pannonian; STE Steppic.
Evidence of sensitivity according to the literature – see text for related reference sources.

Taxonomic group	Species name	Member State report	Evidence in the literature
Mammal (Bat)	*Barbastella barbastellus*	AT, BG, DE, HU, IT, PT	EU
Mammal (Bat)	*Myotis bechsteinii*	BE, BG, DE, HU, IT, PT, SE	EU
Mammal (Bat)	*Myotis nattereri*	BG, DE, HU, SE	EU
Mammal (Bat)	*Nyctalus leisleri*	DE, HU, IT, LT	EU
Mammal (Bat)	*Nyctalus noctula*	BE, DE, HU, IT, LT	EU
Mammal (Bat)	*Plecotus auritus*	DE, HU, IT, PT	EU
Mammal	*Pteromys volans*		FI
Mammal	*Muscardinus avellanarius*		Yes
Mammal	*Lepus timidus*		SE
Reptile	*Coronella austriaca*	BG, LT, RO	
Reptile	*Elaphe longissima*	BG, RO	
Beetle (bark)	*Cucujus cinnaberinus*	AT, CZ, IT, LT, LV, SI, SK	CZ
Beetle (bark)	*Rosalia alpina*	CZ, SI, SK	IT
Beetle (ground)	*Rhysodes sulcatus*	AT, CZ, PL, SI, SK	
Beetle (saproxyllic)	*Cerambyx cerdo*	AT, IT, SK	SE
Beetle (saproxyllic)	*Lucanus cervus*	AT, ES, IT, SK	SE
Beetle (saproxyllic)	*Osmoderma eremita*	CZ, ES, IT, SK	SE
Plant (bryophyte)	*Buxbaumia viridis*	BG, CZ, DE, LV, SE	SE
Plant (bryophyte)	*Dicranum viride*	BG, CZ, HU, LV, PL, RO	SE
Plant (vascular)	*Galanthus nivalis*	HU, IT, RO	

Evidence of impacts of clear cutting on BD birds

The replacement of semi-natural forests with plantations has a particular profound impact on a high proportion of forest species, especially when it results in large-scale even-aged coniferous monocultures (Fuller and Robles, 2018; Roberge, Virkkala and Mönkkönen, 2018). Although the early stages of regrowth can provide suitable scrub-like habitat for some open ground BD bird species, such as Nightjar (*Caprimulgus europaeus*) and Woodlark (*Lulula arborea*), once the forest canopy closes it becomes inhospitable for these species. Some

[89] https://www.naturvardsverket.se/upload/stod-i-miljoarbetet/vagledning/natura-2000/paf-se-mar-2013.pdf

forest species may also benefits from small scale clearance, of closed canopy forests. For example, where this occurred on poor soils it favoured pine and Ericaceae, and created new habitats for Western Capercaillie (*Tetrao urogallus*), Black Grouse (*Lyrurus/Tetrao tetrix*) and sometimes for Hazel Grouse (*Bonasa bonasia*) (Klaus, 1991). This led to an increases of these species over the last centuries, but more modern forestry practices with large-scale clearcutting and reduced rotation time is often detrimental to these species.

However, the effects of clearcutting on Capercaillie can vary, depending on their scale and the indirect effects of other factors, and in some cases seem to be uncertain. For example, a study of the effects of increases in clearcutting in Romania on the Western Capercaillie, found that clearcutting had negative small-scale (lek level) and landscape-scale impacts, with forest areas where 30% was clear-cut showing a reduction in counts of males at leks of 76% (Mikoláš et al, 2015). In contrast low intensity selective logging had a small positive effect. But such impacts of clear cutting were not observed in Norway, where the abundance of adult Capercaillie remained unchanged despite the proportion of old semi-natural forest being halved and replaced by clearcuts and young plantations (Wegge and Rolstad, 2011). This was considered to be due to the species' partial ability to adapt to the new habitat combined with a coinciding reduction in its main predator, the Red Fox. In Finland, a long-term decline in Capercaillie numbers appears to coincide with the increase in intensive forestry, but a statistical analysis of the trends does not support the hypothesis that the decline is due to changes in the forest age structure (Sirkiä et al, 2010). However, the authors could not exclude the possibility that other factors behind the decline may have interacted with forestry in general.

Many other BD bird species are also impacted. For example, the Grey-faced Woodpecker *(Picus canus)* is threatened by changes in forestry practice, including the shortening of the rotation period resulting in the loss of potential nesting trees and a marked reduction in the time-span available for nesting. Therefore, conservation recommendations for the species are that forestry management should be reduced in intensity and the clear cutting of older woodland avoided in order to help build and maintain a network of old deciduous woodland.

Figure 3-24 Birds Directive Annex I species reported as under high pressure from forest clearance

Evidence of sensitivity according to the literature – see text for related reference sources.

Species name	Member State reporting forest clearance pressure	Evidence in the literature
Accipiter gentilis arrigonii	IT	
Aegolius funereus	BG, FI	
Aquila pomarina	BG, LV	
Bonasa bonasia	BG, LV	Yes
Casmerodius albus albus	IT	
Ciconia nigra	PT, SK	
Circaetus gallicus	PT	
Coracias garrulus	LV	
Dendrocopos leucotos	IT, SI, SK	
Dendrocopos medius	AT, BG, LU, LT, SI	
Dryocopus martius	BG, IT, LT, LU, SI	
Egretta garzetta garzetta	IT	
Ficedula albicollis	SI	
Ficedula parva	BG, LV	

Ficedula semitorquata	BG	
Glaucidium passerinum	BG, LT	
Haliaeetus albicilla	BG, LV	
Hieraaetus pennatus	PT	
Milvus migrans	BG	
Milvus milvus	PT	
Pandion haliaetus	LT, LV	
Pernis apivorus	BG, FI, LU, SI	
Picoides tridactylus	AT, BG, LT, LV, SI	
Picus canus	AT, BG	Yes
Strix uralensis	BG	
Lyrurus / Tetrao tetrix		Yes
Tetrao urogallus	BG, FI, LT	Yes

3.4.6 Stump removal and whole-tree harvesting

General biodiversity impacts

Stump lifting for bioenergy purposes has been practiced in Finland and Sweden since the 2000s, including Birch (*Betula* spp.), Norway spruce and Scots pine stumps (Berg, 2014). In Finland, 16% of the total renewable energy from commercial forest chips was produced from stumps and coarse roots in 2013 (Persson, 2017). In other countries stumps are not often removed for commercial purposes (Berg, 2014). Whole tree harvesting (i.e. tree harvesting and removal complete with small branches and leaves) is currently not widely employed for bioenergy production.

Stump lifting is only practiced in forests managed by clear cutting. Stumps are taken from clear cuttings after logging residues have been removed from the site. Stumps and roots are harvested within a year of clear-cutting and piled into heaps on-site (Berg, 2014; Godbold and Walmsley, 2009; Laitila, Ranta and Asikainen, 2008). Soil scarification operations are applied separately.

Evidence of impacts of stump removal on HD habitats

Stumps constitute a unique feature in EU forest habitats and deserve full consideration for its conservation as this practice leads to significant damage in their condition. However, no evidence of impacts on HD habitats as a consequence of stump removal has been found in the literature. It is likely that no forests in which stump removal takes place have Annex I forest habitat characteristics as their nature conservation value is too low and their tree species composition is not natural. The Article 17 reporting format for HD habitats and species does not specifically refer to stump removal as a pressure, considered to be part of forest clearance pressure.

Evidence of impacts of stump removal on HD species

Stumps are key habitats for some HD species, and stump harvesting has been shown to affect saproxylic beetles. It has been shown that stumps support a similar species richness of saproxylic beetles compared to deadwood logs in intensively managed Boreal forests (Jonsell and Hansson, 2011). A study found that if stumps are extracted from a high proportion of the clear-cuts in a region of boreal forest, the current threshold of retaining 15-25% of the stump volume will be insufficient for preserving the beetle fauna in the stumps (Victorsson and Jonsell, 2013b). Landscape-scale data were used to estimate that of 39 analysed saproxylic beetle species in Swedish boreal forest, 13 have more than 10% of their populations in clear-cut stumps, 9 more than 50%, and 5 more than 80% (Jonsell and Schroeder, 2014). Similarly, a study of saproxylic beetle assemblages in oak and pine stumps in French forests confirms their value as habitats for rare species (Brin et al, 2013). Furthermore, extracted stumps stored next to clear-felled sites for two or more years after extraction in boreal forests could result in a trap. A study indicated that the stump storage piles were a severe ecological trap for *Corticaria rubripes*, *Leptusa fumida*, *Dadobia immerse* and *Phloeocharis subtilissima*, which are attracted to the stumps and lay their eggs in them, although other species were not affected and those highlighted are not HD species (Victorsson and Jonsell, 2013a).

A comparison of species richness, abundance and assemblages of saproxylic beetles on low stumps, high stumps and logs of spruce (*Picea abies*) on clear-cuts in northern Sweden (Hjältén, Stenbacka and Andersson, 2010) found a clear difference in species assemblage composition but no significant difference in species richness or abundance between substrates. The authors conclude that low stumps are overlooked as substrates for wood living organisms. A common mitigation method is to retain some high stumps (i.e. leaving 3-5m high stumps of some stems at felling). A simulation of long-term impact (Johansson, Felton and Ranius, 2016) concluded that even at low levels of stump extraction, species which are already rare and specialized on sun-exposed coarse dead wood may be threatened, whilst slash extraction (i.e. woody debris generated as a result of logging operations) probably has less impact, since fewer species are specialized on fine woody debris.

In very intensively managed forests, interventions may need to be more large-scale in order to have a significant impact. A survey of boreal coniferous forest found no differences of understory moss species richness and composition between stump harvested and conventional clear-cut stands 24-36 years after a decade of large scale stump harvesting (Bergmeier, Petermann and Schröder, 2010). Authors concluded that the lack of differences may have been because the additional disturbance effects generated by stump harvest were overridden by the strong disturbance effects already imposed by the standard forest management, i.e. clear-cutting with subsequent soil scarification.

Evidence of impacts of stump removal on BD birds

Many species are potentially impacted from the removal of stumps from forested areas (Roberge, Virkkala and Mönkkönen, 2018). For example, the Redwing *(Turdus iliacus)* and the Common Redstart *(Phoenicurus phoenicurus)* sometimes nest in old stumps. Furthermore, the loss of deadwood and associated invertebrates is likely to be detrimental for a number of species, most obviously woodpeckers, but also some other invertebrate feeders. However, no direct evidence could be found of impacts on any Annex I species.

3.4.7 Tree planting and use of pesticides and fertilisers

General biodiversity impacts

Following clear cutting, trees are usually replaced through large scale planting. To help planting and increase the survival and growth of saplings, mechanical site preparation may be carried out to provide the required soil conditions and hydrology. Site preparation may include scarification (i.e. rearrangement of some of the leaf litter, exposing the mineral soil below), mounding (i.e. piling of material for protection or concealment), and sub-soiling/ripping (i.e. ploughing to a depth below the normal level in order to break up the subsoil), as well as intensive soil preparation methods such as deep ploughing and terracing (Thiffault and Roy, 2011). These processes lead to substantial changes in soil structure and condition and in turn soil fauna.

Tree planting management practices have an overall negative impact on biodiversity. For example, Haeussler (2004) showed that site preparation may alter plant diversity in the short-term, but the impact on biodiversity also depends on its intensity. It has also been highlighted that it takes more than 50 years for nematode diversity to recover following intensive soil preparation (Thornton and Matlack, 2002) and amphibians have been shown to be impacted from intensive site preparation (Hartley, 2002). Various authors have highlighted the need to favour soil preparation methods that reflect natural disturbances and conserve coarse woody debris in order to protect biodiversity (Hartley, 2002; Lof et al, 2012).

Soil preparations and planting may be accompanied by the use of pesticides and fertilisers, which is expected to have considerable implications on biodiversity where the practices are carried out. The use of the insecticide imidacloprid for the control of wood-boring insects in planted trees has been reported to have negative impacts on earthworms and possibly other soil macrofauna (Kreutzweiser et al, 2008).

Evidence of impacts of tree planting practices on HD habitats

The intense planting practices described above would lead to severe damage of all HD forest habitats. Although afforestation can be used as a tool for the restoration of Annex I forest habitats by re-creating forest canopies with the typical species mix of Annex I habitats in previously deforested areas, this is only carried out in a few Natura 2000 sites and has no connection to the use of wood or residues for bioenergy.

Ten Member States (particularly Austria, the Czech Republic, Hungary and the Netherlands) reported 18 HD forest habitats under high pressure from forest replanting, without always specifying if the planting is with native or non-native trees. In addition, Italy and Portugal reported 5 forest habitats as under high pressure from planting of non-native trees on open ground (see Figure 3-8). As discussed below, two Member States (Bulgaria and Germany) specified that the pressure was due to planting with non-native species.

France reported a high pressure from use of biocides in forestry on caves not open to the public in the Continental biogeographic region, and a high pressure from use of fertilisers in forestry on natural dystrophic lakes and ponds in the Atlantic biogeographic region.

Evidence of impacts of tree planting practices on HD species

Eight Member States reported a high pressure of the use of forestry pesticides, other biocides or hormones on 31 HD species, including 13 bat species and 8 saproxylic / bark beetles.

Four Member States reported a high pressure of the use of fertilisers in forestry on 5 HD species / taxa.

Evidence of impacts of tree planting practices on BD birds

Modern forestry practices, including the use of biocides and fertilisers, have been detrimental to Western Capercaillie (*Tetrao urogallus*), Black Grouse (*Lyrurus/Tetrao tetrix*) and Hazel Grouse (*Bonasa bonasia*) (Klaus, 1991).

3.4.8 Forestry with dominance of non-native or non-site typical tree species

General biodiversity impacts

The planting of non-native deciduous and coniferous tree species is regarded as an option for bioenergy production since these species can have higher growth potentials compared with native trees. Introduced non-native tree species occupy around 4.4% of total European forest area according to the latest reports to Forest Europe for SFM Indicator 4.4 (Forest Europe, 2015). Belgium, Denmark, Ireland, and Hungary reported more than 40% of their forest area as planted with introduced species, the Netherlands, Luxembourg and Portugal reported over 25%, according to the Forest Europe report (Forest Europe, 2015). France, Portugal and Spain reported the highest total areas (NB the UK did not report) (Forest Europe, 2015). In Mediterranean countries, plantations of Eucalyptus spp. cover c 1.3 million ha, with more than 80% of this area in the Iberian Peninsula, but also in France and Italy (San-Miguel-Ayanz et al, 2016).

Numerous authors have highlighted the need to favour native tree species over exotics to enhance forest biodiversity, particularly in natural forests (Brockerhoff et al, 2008a; Carnus et al, 2006; Pejchar, Holl and Lockwood, 2005; Wagner and Stephens, 2007). For instance, Schmid, Pautasso and Holdenrieder (2014) showed that fungal diversity is reduced by afforestation of Douglas fir in Europe and that its specific microclimatic conditions can result in reduced arthropod densities in the winter months, which can have a negative impact on birds. Magura, Tóthmérész and Bordán (2000) found that carabid beetles were significantly more abundant, and they exhibited higher diversity in native forest compared with non-native unmanaged spruce plantations in Hungary. The carabid assemblage in the managed spruce plantation was similar to that of the native forest but strikingly different from that in the unmanaged plantation. In Portugal, Proença et al (2010) showed that forest plant and bird richness and diversity were higher in oak forest than in pine and eucalypt plantations. Finch and Szumelda (2007) concluded that afforestation with exotic Douglas fir lead to conspicuous changes in epigaeic invertebrate communities in north-western Germany, including ants, ground beetles and spiders, as compared to autochthonous, deciduous forests of the same age class.

It has also been suggested that native species should also be not only regionally native but also locally native (Bremer and Farley, 2010; Kanowski et al, 2003).

Evidence of impacts of forestry with non-native/non-site typical trees on HD habitats

Two Member States (Bulgaria and Germany) reported a high level pressure from forest replanting with non-native trees under Article 17 (Figure 3-25). It is evident that replacing native tree with non-native trees species can have a devastating effect since the HD habitat itself is eliminated or substantially modified. In fact, the replacement of non-native tree species with locally native species could contribute to the restoration of the conservation status of some Annex I forest habitats.

In Portugal, a study found lower forest plant species richness, diversity and evenness in plantations of native *Pinus pinaster* and non-native *Eucalyptus globulus* compared with patches of natural forest of *Quercus* spp., and some forest species were exclusively observed in the oak forest (Proença et al, 2010), whilst another study found that young and very mature eucalypt plantations may have a similar diversity and composition of understorey vegetation as neighbouring semi-natural scrub (Calviño-Cancela, Rubido-Bará and van Etten, 2012). In section 3.3.4, the impacts of Eucalyptus plantations on freshwater species and habitats are discussed.

Figure 3-25 Annex I forest habitats reported as under high pressure from forest replanting of non-native trees

Key to biogeographical regions: ALP Alpine; ATL Atlantic; BLS Black Sea; BOR Boreal; CON Continental; MAC Macaronesian; MED Mediterranean; PAN Pannonian; STE Steppic.
Evidence of sensitivity according to the literature – see text for related reference sources.

Habitat code	Habitat name	Member State
2180	Wooded dunes of the Atlantic, Continental and Boreal region	BG (BLS)
9120	Atlantic acidophilous beech forests with Ilex and sometimes also Taxus in the shrublayer (Quercion robori-petraeae or Ilici-Fagenion)	DE (ATL)

Evidence of impacts of forestry with non-native/non-site typical trees on HD species

Seven Member States reported 19 HD species as being under high pressure from forest replanting with non-native trees. Another nine Member States reported a high pressure from forest replanting on 15 other HD species without specifying if with non-native or native trees.

15 of the reports come from Spain, and 10 from the Czech Republic. In Spain, this could be an indication of the pressure of *Eucalyptus globulus* plantations, particularly on Lepidoptera.

Figure 3-26 Habitats Directive Annex II, IV and V species reported as under high pressure from forest replanting with non-native species

Key to biogeographical regions: ALP Alpine; ATL Atlantic; BLS Black Sea; BOR Boreal; CON Continental; MAC Macaronesian; MED Mediterranean; PAN Pannonian; STE Steppic.
Evidence of sensitivity according to the literature – see text for related reference sources.

Taxonomic group	Species name	MS reporting high pressure from forest replanting with non-native species	Evidence in the literature
Mammal (bat)	*Barbastella barbastellus*	BE (CON,ATL)	
Arthropod (saproxylic beetle)	*Lucanus cervus*	ES (ATL,ALP,MED), PT (ATL,MED)	
Arthropod (saproxylic beetle)	*Osmoderma eremita*	SK (PAN)	
Arthropod (saproxylic beetle)	*Rhysodes sulcatus*	CZ (CON)	
Arthropod (saproxylic beetle)	*Rosalia alpina*	CZ (CON,PAN)	
Arthropod (saproxylic beetle)	*Xyletinus tremulicola*	EE (BOR)	
Arthropod (Lepidoptera)	*Eriogaster catax*	ES (ALP,ATL)	
Arthropod (Lepidoptera)	*Euphydryas aurinia*	ES (ALP,ATL,MED)	
Arthropod (Lepidoptera)	*Hypodryas maturna*	CZ (CON)	
Arthropod (Lepidoptera)	*Lopinga achine*	CZ (PAN)	
Arthropod (Lepidoptera)	*Parnassius apollo*	ES (MED,ATL,ALP)	
Arthropod (Lepidoptera)	*Parnassius mnemosyne*	CZ (CON,PAN)	
Mollusc	*Vertigo moulinsiana*	ES (MED,ALP)	
Plant (vascular)	*Anarrhinum longipedicelatum*	PT (MED)	
Plant (vascular)	*Culcita macrocarpa*	ES (ATL)	
Plant (vascular)	*Cypripedium calceolus*	CZ (PAN,CON)	
Plant (vascular)	*Echium russicum*	RO (STE)	
Plant (vascular)	*Gladiolus palustris*	CZ (PAN)	
Plant (vascular)	*Thymus villosus ssp. villosus*	PT (MED)	

Evidence of impacts of forestry with non-native/non-site typical trees on BD birds

Non-native tree plantations have resulted in the shortage of large indigenous trees in the lowlands which impacts the breeding sites of the Eastern Imperial Eagle *(Aquila heliacal)*. Afforestation of heathlands, removal of birch stands and the planting of coniferous monocultures is threatening the Black Grouse *(Tetrao/Lyrurus tetrix)* (de Juana and Boesman, 2013). Although the early stages of growth can provide suitable habitat, it becomes unsuitable as the forest matures, as shown in parts of the UK where declines have been strongly linked to forest maturation (Pearce-Higgins et al, 2007).

Several studies have shown that bird species richness and abundance are higher in natural forest patches than in Eucalyptus plantations (Calviño-Cancela, 2013; Proença et al, 2010). Calviño-Cancela (2013) attributed the effect partly to the low presence of phytophagous insects on *Eucalyptus globulus*, but the studies did not cite impacts on species of conservation interest. A study in NW Spain (Goded et al, 2019) found that species richness of both herbs and birds was consistently lower in Eucalyptus plantations compared to native forests. Furthermore, the abundances of bird species characteristic of agricultural, forest, scrubland and other habitats, were all much lower in Eucalyptus plantations than in native forests, and the relative abundance of cavity-nesting forest birds was at least 64% higher in native forests.

The Azores Bullfinch *(Pyrrhula murina)* has been impacted by the widespread clearance of native forest for forestry plantations. The species is entirely absent from areas where alien invasive plant species (especially *Hedychium gardnerianum*, *Clethra arborea* and *Pittosporum undulatum*), have largely overrun the remaining patches of natural vegetation, suppressing the natural fruit, seed and bud food supply (Ceia, Heleno and Ramos, 2009). Further evidence is provided in Figure 3-27.

Figure 3-27 Birds Directive Annex I species reported as under high pressure from forest replanting of non-native trees

Evidence of sensitivity according to the literature – see text for related reference sources.

Species name	MS reporting high pressure of forest replanting of non-native trees	Evidence in the literature
Caprimulgus europaeus	BE	
Dendrocopos leucotos	BG	
Dendrocopos medius	AT	
Lullula arborea	CY	
Pernis apivorus	PT	
Pyrrhula murina	PT	PT
Tetrao tetrix tetrix		UK

3.4.9 Effects of the construction of forest roads and tracks

General biodiversity impacts

Good access is required to enable the more intensive management practices listed above and use of modern large timber harvesting equipment. This therefore requires the creation of a network of roads and forest tracks. Roads of all kinds have seven general effects: mortality from road construction, mortality from collision with vehicles, modification of animal behavior, alteration of the physical environment, alteration of the chemical environment, spread of exotics, and increased use of areas by humans (Trombulak and Frissell, 2000). Public access also facilitates hunting, which causes further disturbance, and in some cases illegal hunting and persecution of some HD species (e.g. large carnivores) and BD birds.

Evidence of impacts of forest roads and tracks on HD habitats

France and Italy reported roads as a high-level pressure on seven HD forest habitats. Romania reported tracks as a high-level pressure on HD forest habitat 91E0 Alluvial forests with Alnus glutinosa and Fraxinus excelsior (Alno-Padion, Alnion incanae, Salicion albae), and Spain on HD forest habitat 9560 Endemic forests with *Juniperus* spp. However, this reporting does not differentiate between roads and tracks constructed for the primary purpose of aiding forestry and other types of road or track construction that pass through forests. Generally, it is clear that roads and tracks lead to direct habitat loss and fragmentation. Habitat degradation in the vicinity is also likely because of hydrological changes.

Evidence of impacts of forest roads and tracks on HD species

A review of the literature of infrastructure impacts, including paved roads, showed that a reduction in mammal populations has been found at distances of a few hundred metres up to several kilometres from infrastructure (Benítez-López, Alkemade and Verweij, 2010). Wolf (*Canis lupus*) activity is significantly lower within 1,000m of paved roads (Capitani et al, 2006; Kaartinen, Kojola and Colpaert, 2005). The location of Wolf pup raising areas in the Italian mountains is primarily determined by distance from villages and paved roads and sites are significantly more frequent inside protected forests than random control sites (Capitani et al, 2006). Another study found that Wolves in Finnish forests avoided roads with at least 250 m distance and buildings with at least 1,000m distance, suggesting that around half of the sparsely populated boreal forest area is subject to reduced use by the Wolf (Kaartinen, Kojola and Colpaert, 2005). However, a study also in Finland found that Wolves used unpaved forest tracks to facilitate movement between areas of preferred habitat, and found no significant difference in hunting success and population growth between a wolf pack with constrained use of home territory due to the high density of primary roads compared to a wolf pack with fewer roads in its territory (Gurarie et al, 2011). There is also evidence showing that Brown Bears (*Ursus arctos*) select deciduous forests away from roads (Güthlin et al, 2011).

However, forest roads and tracks constructed to aid forest management are more often unpaved roads with little to no traffic, and their negative impact on mammals and birds is often minimal, whilst they may have positive population impacts on large mammals by increasing animal movements (Gurarie et al, 2011).

Evidence of impacts of forest roads and tracks on BD birds

Reviews of the global evidence base of the effects of roads indicates that they often have a variety of significant detrimental impacts on birds (Fahrig and Rytwinski, 2009; Kociolek et al, 2011). These result from the direct effects of habitat loss and fragmentation, vehicle-caused mortality, pollution, and poisoning, and the indirect effects of noise, artificial light, barriers to movement, and the creation of edge habitats. The review by Kociolek et al (2011) also concluded that road mortality and traffic noise may typically have the most substantial effects on birds, whereas Summers, Cunnington and Fahrig (2011) conclude that the main cause of reduced bird species richness and density close to roads is traffic mortality. However, the evidence also shows that impacts are greatest for roads with high volumes of traffic, so the impacts of forest roads and tracks with infrequent traffic is likely to be less than found in studies of more typical public roads. Evidence of the effects of roads on forest habitat suitability for birds and forest bird densities in Europe appears to be limited. One study in Sweden suggested that the effects of forest roads on bird communities may be limited or even mixed, as some species favour the open areas and edge habitats that is created (Helldin and Seiler, 2003). However, another study of forest birds in Cyprus found that even unpaved and sporadically used roads can have detrimental effects on the bird communities, especially on vulnerable species (Mammides, Kadis and Coulson, 2015).

It is important to note that most of the studies of the effects of roads on bird populations have focussed on common and generalist species. There is good evidence that many of the threatened and more specialist BD species that are the focus of this study are known to be particularly sensitive to disturbance, both from traffic and especially the increased presence of humans on foot in the habitat, which roads and paths enable. BD birds that are known to be very sensitive to disturbance include large birds of prey and owls, such as Greater Spotted Eagle (*Aquila clanga*) (Meyburg et al, 2015) and Spanish Imperial Eagle (*Aquila adalberti*) (Sánchez, González and Barov, 2008), as well as Capercaillie (*Tetrao urogallus*) (Birdlife International, 2019c) and Black Grouse (*Lyrurus tetrix*) (Birdlife International, 2019a). The impacts of the creation of forest roads, even at relatively low densities, is therefore likely to be significant for these species.

3.4.10 Summary of evidence of sensitivity of EU protected habitats and species to the use of forest biomass

Figure 3-28 Risk factors associated with the use of forest biomass on EU protected habitats and species associated with forests

Risk factor	General effects & biodiversity impacts	Evidence of sensitivity of HD habitats	Evidence of sensitivity of BHD species	Overall sensitivity of HD habitats	Overall sensitivity of BD species	Comments (incl. links to bioenergy)
Selective logging of trees in extensively managed forests	Commercial extraction normally degrades habitat due to reduced density of mature trees, as well as disturbance. Environmentally sensitive selective logging can provide some biodiversity benefits, e.g. creating some open spaces or removing alien species	No evidence found, but scientific evidence indicates that that it can have a significant impact on habitats compared to unlogged forests.	Clear evidence in the literature that bryophytes, bark and saproxylic beetles, as well as terrestrial molluscs and bats are moderately to highly affected by selective logging.	Moderately to highly negative depending on % extracted, methods and forest age and condition; unless required to meet conservation objectives.	Moderately to highly negative depending on % extracted, methods and forest age and condition; unless required to meet conservation objectives.	Overall bioenergy impacts are uncertain, as whilst impacts are mainly moderate, the link to bioenergy use is likely to be mainly indirect and its extent is uncertain.
Thinning of small and intermediate sized successional trees	Creates a more open structure favouring some species and detrimental for others	Limited evidence found. Few MS reporting high impacts on HD habitats from thinning of tree layer	Limited evidence found in the literature regarding the impact of thinning practices on species, showing both positive and negative effects	Low and mostly negative, but beneficial in undermanaged non-natural forest.	Low and mostly negative for a few species, but beneficial in undermanaged non-natural forest.	Overall impacts are likely to be generally low and variable, and the extent to which they are driven by bioenergy use is uncertain.
Removal of dead wood	Significant disruption of the forest ecosystem and community as a high proportion of biodiversity is	Numerous MS reporting evidence of high impacts on HD habitats from the removal of dead wood	Clear evidence in the literature that bryophytes and saproxylic beetles are highly affected	Highly negative	Highly negative, especially for a large proportion of mosses and liverworts, invertebrates and	There is strong evidence that they reduction of deadwood is very damaging, but the extent to which this

Risk factor	General effects & biodiversity impacts	Evidence of sensitivity of HD habitats	Evidence of sensitivity of BHD species	Overall sensitivity of HD habitats	Overall sensitivity of BD species	Comments (incl. links to bioenergy)
	associated with dead wood		by removal of dead wood		specialist forest birds	is driven by bioenergy use is uncertain
Clear cutting of trees to increase production efficiency	Temporary destruction of forest stands and loss of associated species. Followed by scrubland habitat until mature forest is regained. Results in even aged forest, with low structural diversity and reduced biodiversity, especially if carried out at a large-scale. Impacts can be mitigated by retaining some trees.	Clear evidence is found in the literature that clear cutting has a very high impact on HD habitats. Numerous MS reporting this pressure.	Few studies showing the impact of clear cutting on BHD species, but showing clear evidence of the potentially devastating effect on BHD species	Very highly negative – as leads to habitat destruction, and very slow recovery	Highly negative for most species, especially if at large scale; creates temporary habitat for a few species	Impacts are highly detrimental for all habitats and most species, and it is likely that this practice is being partly driven by increased demand for bioenergy
Stump removal and whole-tree harvesting	Major disruption of soils and removal of all significant woody material if whole-tree harvesting – disrupting forest ecosystem and loss of species requiring deadwood and brash	No evidence of the effect of stump removal on HD habitats has been found in the literature despite its high impact	Clear evidence in the literature that bark and saproxylic beetles are highly affected by stump removal	Very highly negative – as leads to major habitat degradation	Highly negative, especially for a large proportion of mosses and liverworts, invertebrates	Impacts are not well documented but highly detrimental for habitats and probably many species, and it is likely that this practice is being partly driven by increased demand for bioenergy
Tree planting and use of pesticides and fertilisers	Major disruption of soils and creation of fast growing even-aged stands with low biodiversity.	No evidence of the effects on HD habitats has been found in the literature although	Some evidence found on the potentially very high impact of soil preparation. Clear	Very highly negative – as leads to habitat destruction, and very slow recovery	Highly negative for most species	Impacts are highly detrimental, and it is likely that this practice is being partly driven by

Risk factor	General effects & biodiversity impacts	Evidence of sensitivity of HD habitats	Evidence of sensitivity of BHD species	Overall sensitivity of HD habitats	Overall sensitivity of BD species	Comments (incl. links to bioenergy)
	Pesticides disrupt food webs, further reducing biodiversity.	these practices can lead to severe damage	evidence found on the negative impact of the use of pesticides on plant, beetle, bat and bird species.			increased demand for bioenergy
Plantation forestry with dominance of non-native or non-site-typical tree species	Results in even aged unnatural mono-cultures of trees with limited associated species, and with low structural diversity, leading to very low biodiversity, especially if carried out at a large-scale.	Few MS reporting evidence of high impacts on HD habitats from the use of non-native trees in forestry but it is evident that the habitat would be replaced.	Clear evidence in the literature for most taxonomic groups, although impact of non-native species on biodiversity can vary greatly depending on the previous land use and the plantation characteristics	Very highly negative – as leads to habitat destruction	Highly negative for almost all species	Overall impacts are highly detrimental and it is likely that this practice is being partly driven by increased demand for bioenergy
Construction of forest roads and tracks to support forest management and harvesting.	Loss of habitat under footprint of road, hydrological disruption from associated drains. Disturbance from construction and operation, especially if paved and open to public, which can also lead to road mortality of some species.	Evidence not found in the literature but effects are evident	Evidence found in relation to several mammals, but mostly from the creation of paved roads	Low from footprint, but moderate impacts can arise from hydrological disruption for some habitat types	No impacts for most species, but low to moderate for some.	Impacts are variable, and probably partly driven by increased demand for bioenergy

4 The assessment of vulnerability of EU protected habitats and species to pressures from bioenergy production

4.1 Methodology

This semi-quantitative assessment focused on the habitats and species that are protected by the EU Nature Directives that are most likely to be impacted by bioenergy feedstock production because they occur in areas that are, or have the potential to be, used for feedstock production that results in direct changes in land use (e.g. changes in crop type or management), or indirect changes in land use as a result of the demand for land for feedstock production. This draws on the results of the assessment of bioenergy feedstock types and their baseline (2015) production quantities and main locations (Figure 2-25), and the projections for their future use in 2030. It also draws on the results of the review of evidence of the sensitivity of EU protected habitats and species to the biophysical effects of bioenergy feedstock production (Chapter 3).

The analysis involved three steps:

1. Screening:

   The selection of habitats that may potentially be directly or indirectly significantly affected by bioenergy production. All aquatic habitats were excluded, because, although they may be affected by bioenergy production (e.g. pollution) impacts on the habitats and species are complex and too difficult to reliably assess within the scope of this study.

   The selection of species that predominantly occur in broad habitats that are most likely to be affected by bioenergy. All fully aquatic species were excluded.

2. Baseline vulnerability assessment: i.e. estimation of the potential exposure of each selected EU protected habitat and species to bioenergy feedstock production and it's identified risk factors combined with an estimation of their sensitivity to each of the risk factors. This was initially carried out assuming no Natura 2000 protected area constraints (or other significant mitigation) and then assuming mitigation through full protection within Natura 2000 areas.
3. Revision of the vulnerability assessment based on bioenergy feedstock production projections for 2030. This changed the estimated exposure of each habitat and species, but it was assumed that sensitivity and Natura 2000 coverage levels would not significantly change.

4.1.1 Screening of species and habitats

The scope of the assessment covered HD habitats (i.e. the 'natural habitats of Community interest' listed in Annex I of the Habitats Directive), HD species (i.e. 'species of Community interest' as listed in Annexes II and/or IV of the Habitats Directive), and BD birds (i.e. bird species listed in Annex I of the Birds Directive) that require special conservation measures. In the following text, the word species is used to refer to the species or subspecies taxa as listed in the directives.

Screening according to MAES ecosystems

From previous assessments and this study, it is clear that the species most likely to be impacted by bioenergy feedstock production are those of arable habitats, grasslands, forests, scrubland and heathland / shrubland. These species were therefore identified and selected using the EEA assignment of EU protected species to MAES ecosystem type, as included in the EU State of Nature Report Annex D (European Commission, 2015b). During this process it was observed that the classification is not always closely aligned to each species' ecology. This is particularly the case for species associated with cropland. Also, it should be noted that many species have more than one ecosystem listed as a preferred habitat, and use many others over the course of its daily / annual cycles. Therefore it is important to note that <u>the assessments only relate to the impacts within the specific habitat in question</u>, and for some species the net impacts of bioenergy production may be the result of complex interactions amongst various bioenergy effects in different habitats.

From the above list, 12 species were excluded because they are currently believed to be extinct in the EU, or their taxonomic status or their EU native status is questioned[90].

As described in Box 3.1, it is very likely that bioenergy feedstock production will impact a range of aquatic habitats and their species, because of indirect effects such as in relation to soil erosion, water runoff and water quality. As such effects and their impacts are diffuse and complex, it is not possible to assess them with a reasonable degree of reliability within the scope of this assessment. Aquatic species were not therefore included in the semi-quantitative assessment, but a reference to the likely general impacts on aquatic ecosystems and species is included in this study's conclusions.

As a result of the screening, the following species that the EEA consider to be predominantly associated with the following ecosystems (as preferred habitat for HD species, or as breeding and/or wintering habitat for BD birds) were selected:

- **585 HD and BD species** for the assessment of bioenergy feedstock production impacts within **agricultural ecosystems** (excluding permanent crops), comprising:
  - Plants 304
  - Invertebrates 76
  - Amphibians 14
  - Reptiles 60
  - Birds 69 (including, Lesser Kestrel (*Falco naumanni*) a clear erroneous omission from the MAES list)
  - Mammals 62

- **216 HD and BD species** for the assessment of bioenergy feedstock production impacts within **forest ecosystems, comprising:**
  - Plants 16
  - Invertebrates 58
  - Amphibians 26

[90] Extinct or probably extinct: *Leiostyla lamellosa, Leiostyla cassida, Capra pyrenaica ssp pyrenaica*. Taxonomic status or native status questionable or disproved - *Capra aegagrus, Ovis gmelini musimon, Ovis orientalis ophion, Chamaeleo chamaeleon, Armeria neglecta, Pyrus magyarica, Sorbus teodori, Cyclamen fatrense, Centaurium rigualii, Scilla lochiae*.

- Reptiles 13
- Birds 46
- Mammals 57

Habitat screening

Annex I habitats were screened according to the observed or potential feasibility of extracting biomass from them, or for their potential for conversion to an ecosystem that could be used for feedstock production. For example, habitats that are especially wet, sandy, rocky, steep or at high altitude are very unlikely to be used for bioenergy feedstock production or have the potential for economically viable conversion and feedstock production. As a result, all coastal habitats, including dunes and saltmarsh, and all rocky habitats and caves were excluded. In addition, as for species, aquatic habitats, were also excluded as the impacts on these are too complex to reliably quantify within the scope of this assessment.

As a result of the screening, 151 HD habitats types were taken forward, comprising:

- Heath & scrub 12
- Sclerophyllous scrub 13
- Grasslands 32
- Bogs, mires & fens 12
- Forests 82

4.1.2 Baseline vulnerability assessment

The vulnerability assessment of the selected EU protected habitats and species built on and adapted the methodology used to systematically and quantitatively assess the impacts of energy technologies on Biodiversity Action Plan priority habitats and species in the UK for Defra (BIO by Deloitte, IEEP and CEH, 2014). The methodology was based on a standard vulnerability assessment approach multiplying estimates of exposure by sensitivity, developed from an earlier study for the UK Joint Nature Conservation Committee (Tucker et al, 2008). For this current study, the vulnerability methodology was adapted to make it semi-quantitative (i.e. less quantitative) to reflect the greater uncertainties and variation in sensitivity and exposure that is expected to occur across the EU.

The assessments were carried out by ecologists in the IEEP study team with sound knowledge of each of the various habitats / species groups[91], drawing on the reviewed evidence. Each assessment was then checked by other appropriate specialists in the wider project team (i.e. birds by BirdLife International, and other species by ecologists in the Arcadis and NIRAS teams).

[91] birds and most mammals by Graham Tucker; bats, invertebrates, reptiles and amphibians by Gustavo Becerra; and plants and habitats by Evelyn Underwood

Assessment of vulnerability to the direct effects of bioenergy feedstock production

The assessment was based on the following **sensitivity risk factors** (and abbreviations used in the analysis tables) that were identified from the literature review that was carried out in the preceding Chapter 3.

- Conventional crops (i.e. food and feed/forage crops)
    1. Conversion of grassland or other semi-natural habitat or fallow to conventional crop (GCC)
    2. Intensification of grassland management (GI)
    3. Intensification of crop production through changes in crop type on existing arable land, principally arable crops to maize (CI)
    4. Removal of residues from crops (straw etc) (RR)

- Bioenergy from non-food crops or other feedstocks on agricultural land or other open land
    5. Conversion of grassland or shrubland to Miscanthus etc or SRC (GBE)
    6. Conversion of arable cropland to Miscanthus etc or SRC (ABE)
    7. Afforestation of grassland or shrubland (GAF)
    8. Afforestation of arable cropland (AAF)
    9. Small-scale planting of trees in farmland (PTF).

- Forest biomass
    10. Selective logging of trees in extensively managed forests (SLOG)
    11. Thinning of small and intermediate sized successional trees (THIN)
    12. Removal of dead wood (RDW)
    13. Clear cutting of trees to increase production efficiency (CCUT)
    14. Stump removal and whole-tree harvesting (SRWTH)
    15. Tree planting and use of pesticides and fertilisers (TPPF)
    16. Conversion of existing forest to plantation forestry (CPF)
    17. Construction of forest tracks to support forest management and harvesting (CFR)

For each EU protected habitat or species, baseline potential vulnerability (PVb) was estimated as the general percentage exposure of the habitat or species to the production of the bioenergy feedstock (E) multiplied by the sum of the percentage sensitivity of the species or habitat to each risk factor (S1, S2 etc) arising from the feedstock's production multiplied by the percentage exposure of the habitat or species to that specific risk factor (ES1, ES2 etc). Thus:

$$PVb = \sum_{i=1}^{n} E\ x\ (S1\ x\ ES1 + S2\ x\ ES2\ \ldots. + Sn\ x\ ESn)$$

The assessment comprised the following key steps described below. Each step was completed for each species or habitat in an Excel spreadsheet (downloadable as a separate file).

Exposure scoring

Baseline exposure is the estimated proportion (as a percentage) of the habitat extent or species population in the EU that is affected by the sensitivity risk factors associated with each of the types of bioenergy feedstock. The baseline exposure is the result of the bioenergy market and technologies as influenced by the implemented policies, legislation and targets in place in EU countries in 2015, as discussed in chapter 2.

For habitats, exposure was based on the area lost or degraded by the physical footprint of each technology. For species, exposure was derived from estimates of the amount of the species' habitat affected. In practice exposed areas may be larger (e.g. as a result of the effects of pollution), but such effects could not be taken into account with this assessment. The baseline exposure estimate was informed by, and cross-checked with, the list of habitats and species reported by Member States (in the Article 12/17 reporting database), and in the wider literature review (chapter 3), as being under pressure over the 2007/8-12 reporting period from biofuel production, or potentially other types of bioenergy feedstock production (e.g. grassland conversion to arable, forest clear felling).

Exposure was assessed in **two** steps:

1. Estimation of general existing or potential exposure to bioenergy feedstock production (**E**), i.e. the % of the EU habitat / species population that occurs in the broad habitats in question (i.e. arable land, grassland and shrubland, or forest) in areas that are being used for bioenergy feedstock production, or have the reasonable potential (including through habitat conversion) to be used for bioenergy feedstock production (excluding consideration of constraints from protected areas designation).
2. Estimation of the % of **E** (i.e. the habitat area with potential for bioenergy feedstock production) that is affected by each sensitivity risk factor (**ES1**, **ES2** …).

Step 1 was assessed for each habitat / species according to the following categories of the EU habitat area or species population potentially affected:

4 = 75 - 100%

3 = 55 - 74%

2 = 25% - 49%

1 = 5 – 24%

0 = <5%

Habitats and species with less than 5% existing or potential exposure (E) were not considered further in the assessment, because their vulnerability would not be potentially significant once the actual exposure component (step 2) is taken in to account.

Step 2 was based on the assessment of the 2015 baseline use of bioenergy feedstock types as set out in chapter 2 (Figure 2.28). Where relevant the impacts relate to the cumulative effect of changes between 2010 and 2015, as 2010 is when bioenergy demand increased significantly as a result of the RED. This provided a general estimation of the overall proportion of the broad habitat types affected by each bioenergy feedstock type, and very importantly, each associated risk factor. Based on this the following 'sensitivity risk exposure' estimates were applied consistently to all habitats and species, in combination with the species specific results of Step 1.

- Conventional crops (i.e. food and feed/forage crops)
    - Conversion of grassland, shrubland or fallow to conventional crop (GCC) = 0.019%
    - Intensification of grassland management (GI) = 0.005%
    - Intensification of cropland through changes in crop type on existing arable land, principally arable crops to maize (CI) = 0.53%
    - Removal of residues from crops (straw etc) (RR) = 0.79%

- Bioenergy from non-food crops or other feedstocks on agricultural land or other open land
    - Conversion of grassland or shrubland to Miscanthus etc or SRC (GBE) = 0.005%
    - Conversion of arable cropland to Miscathus etc or SRC (ABE) = 0.005%
    - Afforestation of grassland or shrubland (GAF) = 0.047%
    - Afforestation of arable cropland (AAF) = 0.002%
    - Small scale planting of trees in farmland (PTF) = 0.005%

The estimation of the exposure of habitats and species to forest biomass sensitivity risk factors also drew on the estimation of the bioenergy feedstock baseline use in chapter 2. In particular it builds on the estimation that 24.8% of primary forest biomass produced in EU forests in 2015 was being used for bioenergy production (Figure 2.28). It is therefore assumed that, in general and on average, the exposure of habitats and species to these risk factors as a result of bioenergy production is proportionate to the amount of forest biomass that is used for bioenergy, which is rounded up to 25%. Thus, for example, if 20% of forests are subject to clear cutting, then approximately 5% of the area is driven by bioenergy feedstock demand.

Similar assumptions were made in estimating the proportional exposure from conventional food crops based on the proportion that is used for bioenergy (Figure 2.28). Clearly such assumptions are crude, and do not take into account the possibility that in the absence of bioenergy demand some biomass would be used for other purposes rather than remaining in the forest ecosystem. Certain forestry practices may also continue to some extent without the use of the products for bioenergy (e.g. thinning and dead wood removal to improve would production in general). On the other hand, bioenergy demand can make the extraction of thinnings and deadwood profitable in some circumstances where it would not otherwise be. As a result of this complexity and the absence of economic data it is not feasible to reliably calculate the baseline use of biomass under such a counterfactual situation. Therefore, whilst the proportion of primary biomass used for bioenergy is taken in this study as a proxy for the proportional exposure to bioenergy related risk factors - the implications of this simplification should be borne in mind when interpreting the results.

The calculation of the exposure of each habitat and species to the forestry related risk factors also requires one further step: the estimation of the average proportion of the EU forest areas that are generally subject to each of the forestry practices that give rise to each of the risk factors. These are set out in Figure 4-1. As these practices vary across regions and forests types and information on these factors is limited (e.g. as some practices are relatively new, such as stump removal and whole tree harvesting) the estimations are very difficult and are mostly approximate assumptions that are used in this study to indicate broad levels of exposure. The results of the forestry related vulnerability assessments in particular should therefore be treated with caution in the light of these data gaps and assumptions.

Figure 4-1 Estimation of exposure of forestry related risk factors within forest habitats in which EU protected habitats and species significantly occur

NB these calculations relate to the exposure of habitats and species to each forestry related sensitivity risk factor within forests other than plantations, as no EU protected habitats occur within plantations and HD and BD species are largely absent.

S	Sensitivity risk factor	Basis for estimate and assumptions	% exposed to risk	% at risk due to 25% bioenergy driver
10	Selective logging of trees in extensively managed forests	Assumed to mainly only occur in areas under conservation through active management (MCPFE category 1.3) which covered 7.6% of forest and other wooded area in 2015 (EFI, 2015). Assumed a small amount elsewhere, so approx. 10%	10%	2.5%
11	Thinning of small and intermediate sized successional trees	Typical practice following clear cutting (see below) and planting, so assumed c. 66% (see clear cutting below).	66%	16%
12	Removal of dead wood	No direct data on the % of forest where the removal of deadwood is significant, but there is extensive evidence in Ch 3 that low deadwood volumes are a pressure on biodiversity (e.g. Johannson et al, 2013). Assumed to be inadequate in 75% of forest	75%	19%
13	Clear cutting of trees to increase production efficiency	Assumed to be normal practice outside areas with selective logging (10% see above) and none and minimum intervention areas (MCPFE categories 1.1 and 1.2) = 4.6% of forest and woodland; so 85.4% of forest. But 19% of forest is plantation where BHD habitats and species are absent, so % at risk is c. 66%	66%	16%
14	Stump removal and whole-tree harvesting	No data, but common practice in SE and FI, which comprise 32% of forest area, but mostly in plantations – so risk to BHD habitats and species low – assumed c. 10%	10%	2.5%
15	Tree planting and use of pesticides and fertilisers	Tree planting occurs on most areas after clear felling (see above), if assume 10% natural regeneration then 66% - 10% = 56%. % affected by pesticides and fertiliser uncertain, but assumed the same.	56%	14%
16	Conversion of existing forest to plantation forestry, i.e. introduced species (all	According to FAO statistics for 2010 (See Annex 2), 415,000 ha of reforestation on forest land per year (no data for FR, GR,	1.3%	0.32%

	planted stands), or intensively managed stands of indigenous species with one or two species, even age class, regular spacing	LU, NL, MT, UK). 0.261% of forest area for these MS, so assumed approx the same for EU and in 2015. Assumed refers to planted forest. So for 2010 to 2015 baseline = 1.3%		
17	Construction of forest roads and tracks to support forest management and harvesting	Typical practice in all semi-natural forests, so assumed in 87% of forest area (EFI, 2015) but zone of disturbance impact is relatively low, so assumed only 5%.	5%	1.25%

Sensitivity scoring of risk factors

As sensitivity is of no consequence if a habitat is not exposed to a development, then sensitivity was not estimated if general exposure (E) was considered to be less than 5%.

Sensitivity (S1, S2 etc) is the estimated change in the condition or extent of habitat Y or population size of species X that is expected if and where it is exposed to each risk factor associated with the bio-physical effects of each type of bioenergy feedstock. Sensitivity was scored against each of the risk factors as listed above, which were identified in the review of evidence of sensitivity assessment (chapter 3) that arise from the production of each bioenergy feedstock: as listed above.

Sensitivity was scored taking into account the degree to which the habitat or species exhibits key risk factors, in the following categories:

-4 = full sensitivity: the habitat is without doubt completely destroyed or the species population extirpated in the affected area (i.e. 100% loss).

-3 = high sensitivity: major changes in habitat condition (e.g. loss of dominant species and profound changes in structure), so no longer qualifies as an Annex I habitat; or substantial declines in species populations (c. >75%) such that they may not be viable in the long-term.

-2 = moderate sensitivity: substantial changes in habitat condition (e.g. loss of species richness and specialist species but dominant species remain and structure remains intact) - such that in unfavourable condition, or detectable declines in species populations (c. 50-75%), but they remain viable.

-1 = low sensitivity: subtle changes in habitat condition are expected but structure and species composition remain intact, or some changes in species population dynamics are expected but overall population change <25%.

0 = not sensitive: no significant changes likely.

+1 = benefit: potential significant improvements in habitat condition or increase in species population expected. As it is very difficult to quantify potential benefits, and this is not the focus of this study, the score is not further sub-divided.

The sensitivity scoring was based on expert judgement and the results of the review of evidence in chapter 3 (including Member State reports on pressures, and scientific literature). Additional information was also used (e.g. the BirdLife International and IUCN databases, and standard reference works) where necessary, especially for habitats and species that appear to be potentially most exposed to bioenergy production, and for which information from the review of evidence was lacking.

Calculation of vulnerability metric

The assessments were carried out separately for bioenergy feedstocks from agricultural land and from forests, as the habitats and species affected differ greatly between them.

The product of the combined calculations described above provided a potential vulnerability score (PVb), i.e. the predicted impact in the absence of mitigation through protected areas (see below). A score of 16 represents the situation where a habitat or species would be potentially lost (through full sensitivity to at least one risk factor) over 75-100% of the assessed habitat area (i.e. arable land, grassland and shrubland, or forest land). In theory, a higher score could occur where a species is highly exposed to several risk factors that would result in its loss. But in most cases, it is of little relevance if there is more than one critical risk. Also, in practice no PVb scores were greater than 16. Therefore, to standardise the score and make it easier to interpret, it was converted to a percentage by dividing by 16 (PVbmax). Thus, for example, a PVb score of -2% indicates that a habitat or species occurrence is predicted to be <u>very approximately</u> 2% lower within the habitat areas in question as a result of bioenergy feedstock production. A hypothetical example of the sensitivity and exposure scores is provided in Figure 4-2.

Figure 4-2 Simplified illustrative examples of Excel assessments of the potential vulnerability of selected EU protected habitats and species to a bioenergy feedstock type

Explanation of scores and quality codes in text above.

Habitat / Species	**Exposure**	**Risk factor sensitivity & exposure**			**PVb / PVbmax**	**% in N2k**	**RVb / RVb max outside N2k**	**ILUC risk**	**Certainty**
		S1	**S2**	**S3**					
	Risk factor (ES1 etc)	10%	0.005%	0.530%					
	Feedstock production area (E)								
A	4	-4	-3	-2	-10.37%	4%	-9.93%	L	3
B	3	-1	1	-2	-2.12%	68%	-0.68%	H	1
C	3	-3	-3	-2	-5.87%	95%	-0.29%	M	2
D	0	NA	NA	NA	0	15%	0	L	

Assessment of residual impacts

It is important to note that the assessment of vulnerability initially does not consider important mitigation measures, such as restrictions on bioenergy feedstock production within protected areas – and is hence referred to as potential baseline vulnerability (PVb). To estimate the residual vulnerability (RVb), the effects of important mitigation measures were taken into account.

In practice, it was only possible to take into account the levels of protection provided by the network of Natura 2000 sites. Whilst other mitigation measures, such as the biofuel sustainability criteria, and FSC accreditation, may also reduce the vulnerability of some habitats and species, the effects of these measures are very difficult to quantify and they are also likely to overlap to some extent with Natura 2000 coverage - thus providing uncertain levels of additional mitigation. The actual level of mitigation is also uncertain because it depends on whether the measure is specifically protecting that habitat or species, which might not be the case (e.g. forest certifications do not necessarily provide the protection required by a particular saproxylic beetle species). Another complication is that some practical protection is afforded EU protected habitats and species outside the Natura 2000 network, such as through the CAP Pillar 1 greening requirement to maintain permanent grassland, although evidence suggests that such measures are fairly weak in terms of protecting such habitats and species (Alliance Environnement and Thünen-Institut, 2017). Although some national protected area designations outside the Natura 2000 network, and other national policies and measures may provide some additional protection, especially for forests (as reviewed in the country case studies), it was not feasible within the scope of this study to incorporate such EU and national mitigation measures into this broad level EU assessment. However, as Natura 2000 coverage of most of the EU protected habitats and species was found to be high (see results), the potential for a significant over-estimation of residual impacts is only possible for a few habitats and species.

The residual baseline vulnerability (RVb) of each selected habitats and species was therefore calculated by multiplying the potential vulnerability (PvB) by the percentage of the habitats and species that is considered to lie outside the Natura 2000 network. This was calculated using the Member State data submitted with the last Article 12 / Article 17 reports (supplied by the ETC-BD). Thus, the residual vulnerability score indicates the vulnerability of the habitat and species assuming that it is fully protected within the Natura 2000 network, and receives no protection outside. It is therefore important to note that this is a crude estimate of residual vulnerability, and whilst it is based on the clear legal protection given to Natura 2000 sites, this does not confer strict protection that prohibits agricultural and forestry activities. Furthermore, there is evidence from Member State Article 12/17 reports that habitat loss and degradation continues within them e.g. as a result of grassland conversion for crop production and the logging of semi-natural forests (see chapter 3).

For some habitats and species, the percentage coverage by the Natura 2000 network is unknown, in which case the habitat or species concerned was not included in the analysis of residual vulnerability. Therefore, it is important to note that the residual vulnerability scores are not based on the full set of habitats and species exposed to bioenergy feedstock production and are not fully comparable with the potential vulnerability scores. For amphibia, reptiles and mammals, more than 50% of the species had missing Natura 2000 coverage data, in both agricultural and forest areas, and therefore residual vulnerability scores were not calculated for them.

Assessment of vulnerability to indirect land use change

There is a significant risk to many EU protected habitats and species associated with indirect landuse change (ILUC) resulting from bioenergy feedstock production in the EU and elsewhere (as the demand for agricultural products is driven within a global marketplace). In the EU, ILUC is particularly likely in areas that are under extensive forms of agricultural or forestry management, and that have the potential for economically viable agricultural improvement and intensification. Similarly, indirect effects could lead to some areas of natural or seminatural non-agricultural habitat being converted to agricultural uses, or being brought into forestry management, where this is practically feasible and economically viable. However, as there is currently insufficient information available to reliably assess the current scale and broad locations of ILUC in the EU, it is not possible to quantify such impacts. Therefore, this component of the study only provided a broad semi-quantitative assessment of each habitat's and species' potential vulnerability to ILUC.

Each of the selected EU protected habitat and species with a general exposure level of >5% was assessed in terms of their potential vulnerability to ILUC, based on their general sensitivity to habitat change and agricultural / forestry intensification, and the proportion of their area or population that occurs in areas that are most at risk of ILUC as a result of demand for the bioenergy feedstocks considered in this study. Thus, sensitive habitats and species with most of their area / population in areas with most potential for agricultural and forestry improvement, such as in semi-improved grasslands or easily convertible semi-natural habitats, and forests would be at high risk. In contrast those with high proportions in areas that are difficult to convert, improve or intensify (e.g. habitats on steep or high ground, or habitats that are very wet, or dry and/or stony, or remote) are probably at a lower risk of ILUC.

These assessments were made on a simple three level scale as below:

1 = low

2 = moderate

3 = high

Reliability ranking

Each sensitivity and exposure assessment was ranked in terms of its reliability based on the amount of supporting evidence, according to the following scale:

3 = High, e.g. based on more than one MS Article 12 / 17 report and/or published reliable evidence for a well know habitat or species.

2 = Moderate, e.g. based on an Article 12 / 17 report and/or published evidence for the specific habitat or species of its sensitivity to the risk factor.

1 = Low, no direct evidence, and little studied habitat or poorly know species, so sensitivity inferred from general knowledge of the habitat's / species' distribution and ecology; and/or Article 12 / 17 reports or published evidence relating to similar species.

4.1.3 Calculation of 2030 vulnerability

As one of the aims of this study is to assess the potential impacts of future developments in the use of bioenergy, the baseline vulnerability assessment estimations, were adjusted using revised estimates of the risk factor exposure levels based on projections for 2030 - as described in chapter 2 and Figure 2-26. The resulting changes in exposure are set out in Figure 4-3 below.

Figure 4-3 Estimated exposure of EU protected habitats and species to bioenergy feedstock production related risk factors in 2030 based on projections for EU bioenergy demand

	Risk factor	2015 Baseline exposure	2030 Projection	% of 2015 baseline
	Conventional crops (i.e. food and feed/forage crops)			
1	Conversion of grassland or other semi-natural habitat or fallow to conventional crop	0.019%	0.019%	Assumed no change
2	Intensification of grassland management	0.005%	0.005%	Assumed no change
3	Intensification on existing arable land (principally arable crops to maize)	0.530%	0.530%	Assumed no change
4	Removal of residues from crops (straw etc)	0.790%	7.100%	899%
	Bioenergy from non-food crops or other feedstocks on agricultural land or other open land			
5	Conversion of grassland and shrubland to Miscanthus etc or SRC	0.0046%	1.100%	23,913%
6	Conversion of arable cropland to Miscathus etc or SRC	0.0046%	1.300%	28,261%
7	Afforestation of grassland and shrubland	0.047%	0.423%	900%
8	Afforestation of arable cropland	0.003%	0.003%	Assumed no change
9	Planting of trees in farmland.	0.005%	0.005%	Assumed no change
	Forest biomass use in bioenergy	24.8%	30.0%	121%
10	Selective logging of trees in extensively managed forests	2.5%	3%	121%
11	Thinning of small and intermediate sized successional trees	16%	19%	121%
12	Removal of dead wood	19%	23%	121%
13	Clear cutting of trees to increase production efficiency	16%	19%	121%
14	Stump removal and whole-tree harvesting	2.5%	3%	121%
15	Tree planting and use of pesticides and fertilisers	14.0%	17%	121%
16	Conversion to plantation forestry with dominance of non-native or non-site-typical tree species	0.32%	0%	121%
17	Construction of forest roads and tracks to support forest management and harvesting	1.25%	2%	121%

For the 2030 scenario it is assumed that the general exposure levels for each habitat and species would remain approximately the same, as there would be no major changes in the main types of bioenergy feedstock and their broad areas of production. It is also assumed that the Natura 2000 coverage would remain approximately the same. This is because the

European Commission have indicated that the terrestrial network is nearly complete[92] . Although some sites are likely to be added for some countries, it should be borne in mind that the impacts of recently observed habitat degradation and losses may result in some areas being removed from the network. However, as there would still be the potential for the restoration of most degraded sites, it is assumed that any decline in the network coverage would be relatively low. Hence, it seems reasonable to assume that the envisaged small increases in the coverage of the network are likely to be offset by small reductions, resulting in no significant predictable net change.

4.2 Results of the assessment of vulnerability of EU protected habitats and species to pressures from bioenergy production

The results of the vulnerability assessment are set out below in relation to the following elements of the analysis:

- the general exposure of the EU protected habitats and species that are predominantly associated with agricultural and forest habitats to the effects of bioenergy feedstock production;
- the sensitivity of the EU protected habitats and species to the risk factors associated with each type of bioenergy feedstock;
- the estimated direct vulnerability of the EU protected habitats and species based on their sensitivity multiplied by their exposure to bioenergy feedstock risk factors; according to:
  - the estimated 2015 bioenergy feedstock production baselines; and
  - the 2030 projections for bioenergy feedstock production.
- the estimated indirect vulnerability of the EU protected habitats and species as a result of indirect land use change (ILUC) impacts from bioenergy feedstock production.

Conclusions from the analysis are then combined with the results of the review of bioenergy feedstock use, and projections (chapter 2) and the sensitivity analysis literature review (chapter 3) in the following chapter.

4.2.1 The general exposure of the EU protected habitats and species to the effects of bioenergy feedstock production

The exposure of EU protected habitats and species to bioenergy feedstock production in agricultural areas

The general exposure of the EU protected habitats and species that are predominantly associated with agriculture habitats to the effects of bioenergy feedstock production is summarised in Figure 4-4. In this study agricultural habitats include arable land, grassland and shrubland (e.g. heathland and Mediterranean maquis), but exclude permanent crops (e.g.

[92] https://www.eea.europa.eu/themes/biodiversity/natura-2000/the-natura-2000-protected-areas-network

fruit, vineyards and olive groves). The analysis provides a very general indication of the extent to which the habitats and species may be potentially exposed to bioenergy feedstock production, because the broad types of habitat that they occur in generally coincide with those with the potential for bioenergy being feedstock production. It is important to note that it takes into account the possibility that habitats may be converted (where it is considered to be practically and economically feasible to do so currently), such as from shrubland to arable land for bioenergy crop production. It does not take into account the possible impacts of indirect land use change, which is discussed later.

The results indicate that of the 69 habitats considered in this assessment, 74% are considered to be significantly exposed to some extent (i.e. more than 5% overlap between occurrence of the habitat and the potential for bioenergy feedstock production). Of these, the majority (43%) have relatively low exposure, as they are predominantly in areas where there is limited potential for bioenergy feedstock production, for example because the areas are on poor soils or are too dry, wet, steep, rocky, high or remote. Furthermore, because all of these habitats are unimproved agriculturally and all have some form of constraint on agricultural production, none are considered to be fully exposed to bioenergy feedstock production effects. Not surprisingly, examination of the exposure of the different types of habitat, reveals that only grasslands are highly exposed to the potential risks of bioenergy feedstock production, i.e. more than 50% overlap.

The situation for EU protected species is rather different, as a relatively high proportion (63%) are considered to have an insignificant level of exposure (i.e. < 5%) to the potential risks of bioenergy production in agricultural areas. This is due to a high proportion occurring in habitats that are unsuitable for bioenergy feedstock production, because they are too dry or at high altitude etc. It is clear that this is particularly the case with plants, and this is not surprising as a high proportion of HD species (i.e. those listed on Annex II) are rare and specialist species, that tend to be restricted to areas that are unsuitable for agricultural production. For example, a high proportion are Alpine species, or associated with very dry or nutrient poor conditions. In contrast, a substantial proportion of the other taxa groups have exposure levels greater than 50%. And, over all taxa, 13% have high levels of exposure, i.e. greater than 75% overlap.

It is important to bear in mind that the majority of the EU protected species considered here will be primarily associated with semi-natural grasslands and shrubland. Therefore these species will only be significantly exposed to the risks of bioenergy production when it involves the conversion of these habitats to intensively managed agricultural habitats. Some BD birds (i.e. those listed on Annex I) do occur in arable habitats, and are therefore potentially exposed to the risks of crop changes and management intensification, but most are dependent on very low intensity arable farming, and therefore have restricted distributions (e.g. to the dry cereal steppelands of Spain and Portugal).

Figure 4-4 The general exposure of the EU protected habitats and species that predominantly occur within agricultural areas to the effects of bioenergy feedstock production in agricultural areas

A) HD forest habitats

Overall general exposure			**Exposure by habitat group**			
Level	**No.**	**%**	**Group**	**Total assessed**	**No. exposure > 50%**	**% >50%**
75 - 100%	0	0.0%	Heath & scrub	12	0	0.0%
50% - 74%	11	15.9%	Sclerophyllous scrub	13	0	0.0%
25% - 49%	10	14.5%	Grasslands	32	11	34.4%
5% – 24%	30	43.5%	Bogs, mires & fens	12	0	0.0%
0 (<5%)	18	26.1%				
Total	69		Total	69	11	
>5%	51	73.9%				

B) BD and HD species that are predominantly associated with agriculture (i.e. arable, grassland and shrubland)

Overall general exposure			**Exposure by taxa**			
Level	**No.**	**%**	**Group**	**Total assessed**	**No. Exposure >50%**	**% >50%**
75 - 100%	77	13.2%	Plants	304	17	5.6%
50% - 74%	47	8.0%	Invertebrates	76	27	35.5%
25% - 49%	55	9.4%	Amphibians	14	3	21.4%
5% – 24%	39	6.7%	Reptiles	60	18	30.0%
0 (<5%)	367	62.7%	Mammals	62	19	58.0%
			Birds	69	40	30.6%
Total	**585**		**Total**	**585**	**124**	
>5%	218	37.3%				

The exposure of EU protected habitats and species to bioenergy feedstock production in forests

Figure 4-5 summarises the general exposure of the EU protected habitats and species that are predominantly associated with forests to the effects of bioenergy feedstock production in forests. Not surprisingly, this indicates that all EU protected forest habitats are significantly exposed to the risk of the effects of bioenergy feedstock production forests (i.e. the production and removal of forest biomass). Furthermore, 45% are considered to be very highly exposed (i.e. greater than 75%). Examination of the exposure levels in relation to forest type indicate that the lower levels of exposure (i.e. below 50%) tend to be associated with Mediterranean forest types in dry conditions or mountain regions. In contrast, 84% of both temperate forest and Mediterranean deciduous forest types have high exposure to bioenergy feedstock production.

As regards EU protected species, 29% are considered to have insignificant levels of exposure (i.e. <5%). This reflects the fact that some forest species are primarily restricted to particular types of forest, a substantial proportion of which are largely unsuitable for forestry, for example because they are in particularly mountainous, rocky, wet or remote areas etc. However, it should be noted that this situation could change, as modern forestry machinery are increasingly able to work in difficult conditions and new areas that were previously cut off from road networks are being connected and opened up for forestry. Amphibians and mammals, and, in particular, invertebrates and birds are currently at most at risk of high levels of exposure.

Figure 4-5 The general exposure of the EU protected habitats and species that predominantly occur within forests to the effects of bioenergy feedstock production in agricultural areas

A) HD Habitats

Overall general exposure			**Exposure by habitat group**			
Level	**No.**	**%**	**Group**	**Total assessed**	**No. exposure > 50%**	**% >50%**
75 - 100%	37	45.1%	Boreal	8	5	62.5%
50% - 74%	23	28.0%	Temperate	38	32	84.2%
25% - 49%	15	18.3%	Mediterranean deciduous	13	11	84.6%
5% – 24%	7	8.5%	Mediterranean sclerophyllous	10	5	50.0%
0 (<5%)	0	0.0%	Temperate mountain coniferous	3	2	66.7%
			Mediterranean and mountain coniferous	10	5	50.0%
Total	82		Total	82	60	
>5%	82	100.0%				

B) BD and HD species that are predominantly associated with forests

Overall general exposure			**Exposure by taxa**			
Level	**No.**	**%**	**Group**	**Total assessed**	**No. exposure > 50%**	**% >50%**
75 - 100%	104	35.0%	Plants	16	7	43.8%
50% - 74%	55	18.5%	Invertebrates	58	45	77.6%
25% - 49%	32	10.8%	Amphibians	26	13	50.0%
5% – 24%	19	6.4%	Reptiles	13	3	23.1%
0 (<5%)	87	29.3%	Birds	46	35	76.1%
			Mammals	57	32	56.1%
Total	297		Total	216	135	
>5%	210	70.7%				

4.2.2 The sensitivity of the EU protected habitats and species to the risk factors associated with each type of bioenergy feedstock

In order to understand the potential impacts of exposure of a habitat or species to bioenergy feedstock production, ie. its vulnerability, it is necessary to know its sensitivity to the biophysical effects of the feedstock production. This section therefore provides an overview of the estimated sensitivities of habitat and species to the main risk factors associated with feedstock production in agricultural areas and forests, as identified in the literature review (chapter 3).

The sensitivity of EU protected habitats and species to bioenergy feedstock production in agricultural areas

The sensitivity of habitats and species was assessed on a scale of -4 to +1 (see section 4.1.2), with negative numbers indicating a detrimental impact. -4 indicates that the habitat or species would be totally lost in the impacted (exposed area), whereas -1 indicates a small impact that would not affect habitat condition or species population levels measurably. Potentially beneficial impacts were scored as +1 as they are particularly difficult to quantify, and the main aim of this study is to identify risks to EU protected habitats and species so that they can be mitigated.

Figure 4-6 provides a breakdown of the estimated levels of sensitivity of EU protected habitats and species to risk factors associated with bioenergy feedstock production in agricultural areas. As habitats and species that had an exposure level assessment of <5% were not scored for sensitivity, (as it is of no consequence) these are not included in the sensitivity calculations. Sub-tables a) and b) show the breakdown of sensitivity scores, whilst sub-tables c) shows the % of sensitivity scores of -3 or -4 that occurred in relation to each species group, and their mean scores. The scores are not broken down according to habitat groups as they were entirely consistent across all types.

The results clearly indicate that most species are sensitive to the bioenergy feedstock production risk factors in agricultural areas, with the vast majority expected to suffer negative impacts. Furthermore, sensitivity levels are high to very high (i.e. -3 and -4 respectively) for most of the risk factors. This is especially apparent in relation to the conversion of grasslands to cropland, bioenergy crops or forests; each effectively destroying the habitat of the species concerned. The sensitivity of species to impacts on arable land are lower, as many of the species concerned do not occur there and are therefore unaffected. However, for those species that are present, the conversion of the habitat to bioenergy crops or forests also makes the habitat unsuitable for the arable species.

The only risk factors that appear to be of low concern are the removal of residues and the low density planting of trees. Only a couple of BD birds are likely to be sensitive to the removal of residues, primarily as a result of the loss of food resources. However, it should be borne in mind that the removal of residues may have broader impacts on common farmland species and the reduced levels of organic input into the soil may result in wider ecosystem impacts (Ponge et al, 2013; Thiele-Bruhn et al, 2012). The assessment of the sensitivity of species to planting trees on farmland was difficult as the impacts are likely to depend on the types of trees that are planted, their density and the scale of planting over the landscape, and the situation before planting. Thus in some cases where tree densities are artificially low in the landscape, then planting could create benefits were range of species, although these are most likely to be common species, rather than EU protected species. On the other hand, some EU protected species of very sensitive to changes in landscape characteristics (e.g. large flying birds such as bustards and geese). Therefore, the introduction of trees into currently open landscapes would probably be highly detrimental to them.

There appears to be relatively little variation in the sensitivity of the different taxa groups to the risk factors. However, plants have the highest mean sensitivity, whilst mammals the lowest, but given the difficulties in assessing the sensitivity of these different taxa groups, and the nature of the data, it is not appropriate to test their statistical significance.

Figure 4-6 The % levels of sensitivity of EU protected habitats and species to risk factors associated with bioenergy feedstock production in agricultural areas

Sensitivity factors: GCC = Conversion of grassland etc to conventional crops; GI = Intensification of grassland; CI = Intensification on existing arable land; RR = Removal of residues; GBE = Conversion of grassland etc to Miscanthus etc; ABE = Conversion of arable land to Miscathus etc; GAF = Afforestation of grassland etc; AAF = Afforestation of arable land; PTF Planting of trees in farmland.

a) All HD habitats combined

Sensitivity level	GC	GI	CI	RR	GBE	ABE	GAF	AFF	PTF
-4	100%	100%	0%	0%	100%	0%	0%	0%	0%
-3	0%	0%	0%	0%	0%	0%	0%	0%	20%
-2	0%	0%	0%	0%	0%	0%	0%	0%	80%
-1	0%	0%	0%	0%	0%	0%	0%	0%	0%
0 (<5%)	0%	0%	100%	100%	0%	100%	100%	100%	0%
+1	0%	0%	0%	0%	0%	0%	0%	0%	0%

b) All BD and HD species combined

Sensitivity level	Risk factor	GI	CI	RR	GBE	ABE	GAF	AFF	PTF
-4	60%	39%	6%	0%	63%	13%	62%	15%	3%
-3	23%	50%	10%	0%	14%	6%	21%	5%	2%
-2	10%	6%	13%	0%	10%	9%	11%	11%	3%
-1	2%	1%	6%	4%	1%	5%	1%	4%	5%
0	4%	3%	65%	95%	11%	65%	3%	57%	60%
+1	0%	1%	0%	0%	0%	2%	1%	9%	27%

c) According to taxa group, for sensitivity >=-3

Taxa group	GC	GI	CI	RR	GBE	ABE	GAF	AFF	PTF	Mean sensitivity
Plants	87%	92%	16%	3%	92%	21%	87%	21%	16%	-1.92
Invertebrates	96%	100%	11%	0%	100%	13%	98%	13%	0%	-1.08
Amphibians	100%	100%	10%	0%	90%	30%	90%	0%	0%	-1.34
Reptiles	86%	82%	29%	0%	96%	29%	96%	14%	0%	-0.91
Birds	76%	87%	24%	0%	71%	27%	93%	40%	9%	-1.53
Mammals	79%	83%	14%	2%	83%	19%	79%	19%	14%	-0.77

As all HD habitats, including those that occur within agricultural areas, are semi-natural habitats that have very specific ecological requirements, all habitats are sensitive to the bioenergy risk factors. This is clearly the case with respect to all the risk factors other than those associated with arable land, where the zero sensitivities or due to the habitats being absent in such areas.

The sensitivity of EU protected habitats and species to bioenergy feedstock production in forests

Figure 4-7 provides a breakdown of the estimated levels of sensitivity of EU protected habitats and species to risk factors associated with bioenergy feedstock production forests. These were carried out in the same way as for agriculture, and do not include habitats and species with <5% general exposure to bioenergy production in forests.

Given their particular character and relatively specific ecological requirements, all the HD forest habitat types are considered to be highly sensitive to all the forestry related bioenergy feedstock production risk factors, with the exception of the creation of roads. Strictly speaking, roads also destroy the habitat within the footprint, and can lead to wider habitat degradation (e.g. hydrological impacts where roads are combined with drains), but their sensitivity is scored lower here due to the relatively small area directly affected by roads.

Figure 4-7 The % levels of sensitivity of EU protected habitats and species to risk factors associated with bioenergy feedstock production in forests

Sensitivity factors: Slog = Selective logging of trees; Thin = Thinning of trees; RDW = Removal of dead wood; CCut = Clear cutting of trees; SRWTH = Stump removal and whole-tree harvesting; TPPF = Tree planting and use of pesticides and fertilisers; CPF = Conversion to plantation forestry; CFR = Construction of forest roads.

a) All habitats combined

Sensitivity level	SLog	Thin	RDW	CCut	SRWTH	TTPF	CPF	CFR
-4	0%	0%	0%	100%	100%	100%	100%	0%
-3	94%	95%	100%	0%	0%	0%	0%	0%
-2	5%	0%	0%	0%	0%	0%	0%	0%
-1	0%	0%	0%	0%	0%	0%	0%	100%
0 (<5%)	1%	5%	0%	0%	0%	0%	0%	0%
+1	0%	0%	0%	0%	0%	0%	0%	0%

b) All species combined

Sensitivity level	SLog	Thin	RDW	CCut	SRWTH	TTPF	CPF	CFR
-4	13%	7%	14%	58%	50%	59%	62%	0%
-3	12%	21%	21%	27%	13%	11%	27%	4%
-2	19%	14%	23%	6%	8%	9%	8%	19%
-1	12%	12%	15%	3%	10%	3%	1%	28%
0 (<5%)	31%	30%	27%	5%	18%	19%	2%	49%
+1	12%	16%	0%	1%	0%	0%	0%	0%

c) According to taxa group, for sensitivity >=-3 [to note that there are some high anomalies that appear to remain in relation to plant ratings, to be checked in parallel to Commission review]

Taxa group	SLog	Thin	RDW	CCut	SRWTH	TTPF	CPF	CFR	Mean sensitivity
Plants	67%	58%	67%	100%	100%	100%	92%	0%	-2.34
Invertebrates	48%	56%	48%	92%	88%	100%	88%	0%	-2.28
Amphibians	8%	8%	8%	92%	96%	96%	96%	4%	-2.17
Reptiles	0%	0%	0%	100%	25%	100%	100%	0%	-0.62

Taxa group	SLog	Thin	RDW	CCut	SRWTH	TTPF	CPF	CFR	Mean sensitivity
Birds	21%	29%	40%	83%	2%	2%	93%	14%	-1.31
Mammals	7%	11%	50%	65%	50%	57%	83%	4%	-1.49

A key result to note from the species assessments are the very high levels of sensitivity to the conversion of forest to plantation, which leads to the potential loss of 62% of species and very high impacts on another 27%. This sensitivity is consistent across all taxa groups. The clearcutting of forest, and to lesser extent tree planting of forest combined with pesticides and fertiliser use, is also highly detrimental for most species. Some species, most of which are birds, may benefit from the creation of openings in closed forests that are created by clearcutting, such as Black Grouse (*Lyrurus / Tetrao tetrix*). However, such benefits are often temporary (until canopy closure), and also highly dependent on them being of an appropriate scale. The sensitivity of species to stump removal and whole tree harvesting is difficult to assess for some taxa, such as birds, but it is certainly considered to be highly detrimental for many, particularly invertebrates. Such invertebrate species are also highly sensitive to the removal of dead wood, which in turn has detrimental impacts on species that depend on them as food resources (e.g. woodpeckers and other invertebrate feeding birds). The loss of old dead trees also reduces the availability of holes for nesting in, which by itself can lead to otherwise suitable habitats being unsuitable for some species (e.g. woodpeckers, owls, and most bats). Consequently, there is good evidence that current deadwood quantities in commercial forests are far too low to maintain the biodiversity dependent on this habitat (Mueller and Bütler, 2010).

The impacts of the creation of forest roads clearly varies considerably depending on the taxa concerned. Impacts on plants and invertebrates are generally low, as they are primarily the result of the footprint of the road itself, which is relatively very small. In contrast, a significant number of birds and a few mammal species, are highly susceptible to disturbance, and therefore forest road impacts (including the indirect effects of increasing human presence in forest areas) extend over a much greater area, and thus species are more sensitive to their presence.

Selective logging of forests and the thinning stands of trees are considered to have slightly lower impacts than the other forest management activities reviewed here, but a sizeable proportion of species are highly sensitive to them. However, the impacts of logging and thinning depend a great deal on their intensity and scale, and if carried out sensitively they can be beneficial for some species. Some types of forest in Europe have arisen as the result of many centuries of traditional management, notably the coppicing and pollarding of trees (Kirby and Watkins, 2015). In fact, across Europe, more than a third of the forest habitat types protected by the Habitats Directive have been, or still are, managed by coppicing or pollarding in some countries, although there is no overall consensus on how essential this is for their conservation (Mairota et al, 2016). However, there is evidence that some Annex II species actively benefit from coppicing in at least some regions, notably the Hazel Grouse (*Bonasia bonasia*) (Mairota et al, 2016), and the Lepidoptera *Eriogaster catax* and *Euphydryas maturna* in Germany (Dolek, Kőrösi and Freese-Hager, 2018), and the Lepidopteran *Euplagia quadripunctaria* in Italy (Greco et al, 2018).

4.2.3 The estimated vulnerability of the EU protected habitats and species based on their sensitivity multiplied by their exposure to bioenergy feedstock risk factors

On the basis of the estimates of general exposure of habitats and species to bioenergy feedstock production (4.2.1), the preceding assessment of their sensitivity to feedstock production risk factors, and the exposure of habitats and species to the specific risk factors (as estimated above, based on the chapter 2 review), the overall vulnerability of each habitat and species was calculated. This only takes into account the direct impacts from bioenergy production (e.g. land use change or land management intensification to produce bioenergy feedstocks), and not indirect land use change (ILUC), which is discussed further below.

The results of the assessment are set out below, based on:

1. The baseline estimates of bioenergy feedstock production and their contribution to the exposure of the habitats and species to each risk factor (see Figure 2-25):
    a. Potential vulnerability, i.e. the vulnerability of each habitat and species, in the absence of key mitigation measures such as restrictions on agricultural and forestry activities within Natura 2000 sites and other protected areas.
    b. Residual vulnerability, which takes into account the percentage coverage of the habitat or species by the Natura 2000 network (based on Member State reporting data), e.g. assuming 100% coverage results in 100% protection and therefore 0% impact (although as discussed in section 3.2.2, this is currently not the case in practice).

2. The 2030 projections for bioenergy feedstock production and their contribution to the exposure of the habitats and species to each risk factor (see Figure 2-26):
    a. Potential vulnerability (as above).
    b. Residual vulnerability (as above).

Important note: When interpreting the results of the vulnerability assessment, it is very important to bear in mind its overall purpose, which is to identify the main types of risk to EU protected habitats and species, and establish which are most likely to have significant impacts that would affect their conservation status. Therefore, whilst the assessment has been carried out for individual habitats and species, the individual results should be interpreted with considerable care. This is due to the considerable difficulties in accurately and precisely estimating the exposure of the habitats and species to the risk factors, and their sensitivity to them. For these reasons it is suggested that **the individual results should not be used as a source of reference for their estimated vulnerability**. Instead, the combined results are presented below to provide a broader picture of the risks of bioenergy production to EU protected habitats and species, and to quantify these in orders of magnitude, rather than in precise figures.

The vulnerability of EU protected habitats and species to bioenergy feedstock production in agricultural habitats

The key finding from the analysis of the vulnerability of EU protected species and habitats to the combined impacts of the risk factors resulting from bioenergy feedstock production in agricultural habitats is that the **baseline 2015 potential impacts of bioenergy feedstock production in agricultural areas are estimated to be very low** (Figure 4-8). This is particularly the case with habitats, with the majority of habitat types having impact scores of less than -0.1%, which very approximately indicates that the habitats concerned have declined quantity and/or quality by about 0.1% as a result of bioenergy production in 2015. Over all habitat types the mean score is -0.009%, or -0.013% when only those habitats that significantly occur in areas that are used for bioenergy feedstock production, or have the potential to be used for this, are taken into account. Clearly these impacts are very small indeed, and unlikely to affect their overall conservation status of the habitats concerned (although local impacts could be significant). Furthermore, there is a relatively high coverage of the habitats within Natura 2000 sites, and therefore the estimated residual vulnerability is also much lower, i.e. -0.001%. Although this does assume full protection within sites, which in practice is unlikely. Similar results are evident for BD and HD species, both in terms of their potential vulnerability and their residual vulnerability once protection within Natura 2000 sites is taken into account (although this could not be reliably calculated for amphibians, reptiles and mammals).

The principal reason for the very low potential vulnerability scores, for both habitats and species, is their very low levels of estimated exposure to the specific risk factors within agricultural habitats that is the result of bioenergy demand, most of which are below 0.01% (Figure 4-3). Therefore, although as indicated earlier in this chapter, a large proportion of habitats and species have high levels of potential general exposure to bioenergy feedstock production, and most of these have high levels of sensitivity to most of the risk factors, once multiplied by the exposure to the specific risk factors, final vulnerability is very low. Although, the risk factor exposure levels are much higher for the intensification on existing arable land at 0.530%, and for the removal of residues from crops at 0.790%, most EU protected habitats and species have low sensitivity to these risk factors.

The 2030 projections for bioenergy production in agricultural areas suggest that there will be little change in the use of conventional crops, but there will be substantial expansion in the production of dedicated bioenergy crops (i.e. Miscanthus and SRC), leading to grassland conversion exposure levels of 1.1% and 1.3% for arable conversion (compared to 0.0046% for both in 2015). Despite these increases, of several thousand percent, the overall vulnerability scores of the EU protected habitats and species are not expected to increase to significant levels overall (the highest vulnerability score for habitats being -0.537% for grasslands, and the highest species group score being -1.26% for birds).

Figure 4-8 The estimated vulnerability of EU protected habitats and species to risk factors associated with bioenergy feedstock production in agricultural areas

a) HD habitats

Combined

	2015 baselines				2030 projections			
	Potential vulnerability		Residual vulnerability (outside N2k)		Potential vulnerability		Residual vulnerability (outside N2k)	
	No.	%	No.	%	No.	%	No.	%
Detrimental Impacts Number of very high impacts (<-50%)	0	0.0%	0	0.0%	0	0.0%	0	0.0%
Number of high impacts (>-10 to -50%)	0	0.0%	0	0.0%	0	0.0%	0	0.0%
Number of moderate impacts (>-1 to -10%)	0	0.0%	0	0.0%	0	0.0%	0	0.0%
Number of low impacts (>-0.1 to -1%)	0	0.0%	0	0.0%	51	73.9%	5	7.4%
Number of very low impacts (0.0 to -0.1%)	51	73.9%	36	52.9%	0	0.0%	31	45.6%
Number with no impact	18	26.1%	32	47.1%	18	26.1%	32	47.1%
Number with beneficial impacts	0	0.0%	0	0.0%	0	0.0%	0	0.0%
Mean score	-0.009%		- 0.001%		-0.339%		- 0.027%	
Mean score where general exposure >0	-0.013%		- 0.001%		-0.458%		- 0.036%	

Mean vulnerability score by habitat group

	2015 baselines		2030 projections	
	Potential vulnerability	Residual vulnerability (outside N2k)	Potential vulnerability	Residual vulnerability (outside N2k)
Heath & scrub	-0.0065%	-0.0002%	-0.2347%	-0.0060%
Sclerophyllous scrub	-0.0054%	-0.0002%	-0.1950%	-0.0063%
Grasslands	-0.0150%	-0.0014%	-0.5370%	-0.0514%
Bogs, mires & fens	-0.0020%	-0.0002%	-0.0705%	-0.0053%

a) HD and BD species that are predominantly associated with agricultural habitats

Combined

	2015 baselines				2030 projections			
	Potential vulnerability		Residual vulnerability (outside N2k)		Potential vulnerability		Residual vulnerability (outside N2k)	
	No.	%	No.	%	No.	%	No.	%
Detrimental Impacts								
Number of very high impacts (<-50%)	0	0.0%	0	0.0%	0	0.0%	0	0.0%
Number of high impacts (>-10 to -50%)	0	0.0%	0	0.0%	0	0.0%	0	0.0%
Number of moderate impacts (>-1 to -10%)	1	0.2%	0	0.0%	123	21.1%	44	12.8%
Number of low impacts (>-0.1 to -1%)	69	11.8%	42	12.2%	86	14.7%	65	18.8%
Number of very low impacts (0.0 to -0.1%)	145	24.9%	83	24.1%	7	1.2%	16	4.6%
Number with no impact	368	63.1%	220	63.8%	368	63.0%	220	63.8%
Number with beneficial impacts	0	0.0%	0	0.0%	0	0.0%	0	0.0%
Mean score	-0.05%		-0.04%		-0.48%		-0.33%	
Mean score where general exposure >0	-0.14%		-0.04%		-1.29%		-0.33%	

Mean vulnerability score by taxa group

	2015 baselines		2030 projections	
	Potential vulnerability	Residual vulnerability (outside N2k)	Potential vulnerability	Residual vulnerability (outside N2k)
Plants	-0.02%	-0.01%	-0.16%	-0.10%
Invertebrates	-0.07%	-0.04%	-0.84%	-0.41%
Amphibians	-0.10%	?	-0.77%	?
Reptiles	-0.08%	?	-0.71%	?
Birds	-0.18%	-0.13%	-1.26%	-0.93%
Mammals	-0.05%	?	-0.44%	?

The Natura 2000 coverage of the agricultural HD habitats considered here is very high, with 91% having over coverage 75%, and the remaining habitats having 50% - 75% coverage (with missing data for one habitat type). Therefore, the estimated residual vulnerability scores under the 2015 baseline and 2030 projections (assuming no significant change in Natura 2000 coverage) are much lower than the potential scores. The Natura 2000 coverage of agricultural HD and BD species is also substantial, although not as high as for habitats. For those species with coverage data, 45% currently have more than 75% coverage, and 14% have 50 – 75% coverage. However, coverage data were missing for 237 species (68%). Therefore, whilst it is evident that the residual scores should be lower than the potential scores, the actual estimates should be treated with some caution. Most of the missing coverage scores were for amphibians, reptiles and mammals, and therefore residual impacts were not separately estimated for these taxa.

It is important to note that that potential scores do not take into account any mitigation of conventional agricultural and forestry practices whilst the residual scores assume full protection with Natura 2000 sites. Clearly this assumption is simplistic, as evidence from Member State reporting and other literature (reviewed in chapter 3) show that agricultural and forestry practices that are detrimental to EU protected habitats and species are carried out. On the other hand, other mitigation measures (e.g. CAP measures, forest certification schemes and others reviewed in this study – see case studies reports) are not taken into account in the estimation of residual vulnerability scores. In reality, the true vulnerability scores probably lie somewhere between the estimates of potential and residual vulnerability. However, they would be closer to the residual scores for 2030 if by then Natura 2000 sites are being more effectively protected and managed in accordance with their conservation objectives.

It is also very important to bear in mind that this general vulnerability assessment almost certainly underestimates the full impacts of current and future EU bioenergy feedstock production on EU protected species and habitats. This is because the majority of them occur in extensively managed semi-natural grasslands and shrublands (with some are dependent on low intensity arable habitats) and these are all disproportionately likely to be converted to bioenergy feedstock production (see chapter 2). Furthermore, the results of the assessment of the vulnerability of the habitats and species to ILUC suggest that this will lead to additional, and potentially more significant impacts (Figure 4-9). Although a large proportion of some habitats (bogs etc) and some species (primarily plants) are considered to have low vulnerability to ILUC (mainly because they occur in particularly unsuitable areas for agricultural production) a substantial proportion of others have a medium or high level of vulnerability. This finding is also generally in accordance with conclusions from other studies, such as Hellman and Verburg (2010), who found that the 'indirect effects of the RED on land use and biodiversity are much larger than its direct effects' (although the study did not take into account mitigation measures, such as in relation to protected areas).

It must also be stressed that, despite the predicted relatively low overall level of impact in 2030, it can be expected that more significant local land use changes will occur, particularly in countries and regions where bioenergy use is much higher than the average (e.g. Germany with respect to maize production). This could lead to larger scale regional impacts that could have an effect on the conservation status of the affected habitats and species.

Figure 4-9 Estimates of the vulnerability of EU protected habitats and species to indirect land use change in agricultural areas as a result of biofuel feedstock demand

NB Excludes consideration of habitats and species with zero general exposure in agricultural habitats

HD habitats

	ILUC vulnerability		
Group	**Low**	**Medium**	**High**
Heath & scrub	0%	100%	0%
Sclerophyllous scrub	13%	88%	0%

Grasslands	23%	58%	19%
Bogs, mires & fens	0%	100%	0%
Overall	**16%**	**73%**	**12%**

BD and HD species that are predominantly associated with agricultural habitats

	ILUC vulnerability		
Group	**Low**	**Medium**	**High**
Plants	24%	50%	26%
Invertebrates	7%	87%	7%
Amphibians	20%	80%	0%
Reptiles	21%	79%	0%
Birds	51%	29%	20%
Mammals	52%	45%	2%
Overall	**32%**	**56%**	**11%**

The vulnerability of EU protected habitats and species to bioenergy feedstock production in forests

In contrast to the findings of the assessment of bioenergy feedstocks in agricultural areas, the vulnerability assessment strongly indicates that bioenergy feedstock production in forests is already having substantial impacts on EU protected habitats and species (Figure 4-10). According to the findings, the vulnerability scores for all habitats are in the high category (i.e. -10 to -50%), and 43% are over -50%. The mean vulnerability score is -47%, and all forest types are similarly affected.

The results for the majority of EU protected species are similar to those for habitats, although there are a number that are considered to be unlikely to be impacted, as the sensitivity to forestry risk factors varies between species. Nevertheless, although the impacts may be lower, the mean level is high at -23%, 50% of the assessed species are considered to be highly vulnerable, and a further 12% to be subject to very high impacts. Invertebrate species are especially vulnerable, as many are highly sensitive to the removal of dead wood, stump removal, and the use of pesticides and invertebrates etc, as well as (like other species) the conversion of forests to plantations.

The high vulnerability scores are due to a combination of the high levels of general exposure to bioenergy feedstock production and forests (as most forests have the potential to be used for such purposes), the high-sensitivity of many EU protected habitats and species to forestry activities (especially removal of dead wood, clearcutting and replanting of forests and the conversion of forests to plantations) and the relatively high proportions of forests that are affected by these practices. Very importantly, according to the analysis of baseline use of bioenergy (Figure 2-25), approximately 25% of the demand for forest biomass can be attributed to the demand for bioenergy use in 2015. Therefore, although there is much uncertainty over the actual levels of exposure and sensitivity of many of the assessed habitats and species, given the magnitude of the vulnerability scores, it can be said with a high level of certainty that currently bioenergy feedstock production in forests is having a substantial impact on the conservation status of a large number of EU protected habitats and species.

Figure 4-10 The estimated vulnerability of EU protected habitats and species to risk factors associated with bioenergy feedstock production in forests

a) Forest habitats

Combined

	2015 baseline				2030 projection			
	Potential vulnerability		Residual vulnerability (outside N2k)		Potential vulnerability		Residual vulnerability (outside N2k)	
	No.	%	No.	%	No.	%	No.	%
Detrimental Impacts								
Number of very high impacts (<-50%)	35	42.7%	0	0.0%	58	70.7 %	0	0.0%
Number of high impacts (>-10 to -50%)	47	57.3%	0	0.0%	24	29.3 %	0	0.0%
Number of moderate impacts (>-1 to -10%)	0	0.0%	1	1.3%	0	0.0%	1	1.3%
Number of low impacts (>-0.1 to -1%)	0	0.0%	78	98.7 %	0	0.0%	78	98.7 %
Number of very low impacts (0.0 to -0.1%)	0	0.0%	0	0.0%	0	0.0%	0	0.0%
Number with no impact	0	0.0%	0	0.0%	0	0.0%	0	0.0%
Number with beneficial impacts	0	0.0%	0	0.0%	0	0.0%	0	0.0%
Mean score	-46.8%		-0.49%		-56.7%		-0.59%	
Mean score where general exposure >0	-46.8%		-0.49%		-56.7%		-0.59%	

Mean vulnerability score by habitat group

	2015 baselines		2030 projections	
	Potential vulnerability	Residual vulnerability (outside N2k)	Potential vulnerability	Residual vulnerability (outside N2k)
Boreal	-41.0%	-0.43%	-49.6%	-0.51%
Temperate	-50.5%	-0.51%	-61.1%	-0.62%
Mediterranean deciduous	-52.2%	-0.58%	-63.2%	-0.70%
Mediterranean scleropyhllous	-39.8%	-0.43%	-48.2%	-0.52%
Temperate mountain coniferous	-40.8%	-0.41%	-49.4%	-0.49%
Mediterranean and mountain coniferous	-39.8%	-0.42%	-48.2%	-0.50%

a) BD and HD species predominantly associated with forests

Combined

	2015 baselines				2030 projections			
	Potential vulnerability		Residual vulnerability (outside N2k)		Potential vulnerability		Residual vulnerability (outside N2k)	
	No.	%	No.	%	No.	%	No.	%
Detrimental Impacts								
Number of very high impacts (<-50%)	35	11.8%	1	0.5%	65	21.9%	7	3.8%
Number of high impacts (>-10 to -50%)	148	49.8%	91	49.5%	126	42.4%	92	50.0%
Number of moderate impacts (>-1 to -10%)	23	7.7%	28	15.2%	15	5.1%	21	11.4%
Number of low impacts (>-0.1 to -1%)	3	1.0%	3	1.6%	3	1.0%	3	1.6%
Number of very low impacts (0.0 to -0.1%)	0	0.0%	4	2.2%	0	0.0%	4	2.2%
Number with no impact	87	29.3%	56	30.4%	87	29.3%	56	30.4%
Number with beneficial impacts	1	0.3%	1	0.5%	1	0.3%	1	0.5%
Mean score	-22.7%		-12.9%		-27.5%		-15.5%	
Mean score where general exposure >0	-32.1%		-17.4%		-38.9%		-21.0%	

Mean vulnerability score by habitat group

	2015 Baseline		2030 projection	
	Potential vulnerability	Residual vulnerability (outside N2k)	Potential vulnerability	Residual vulnerability (outside N2k)
Plants	-29.4%	-16.1%	-35.5%	-19.5%
Invertebrates	-40.8%	-22.1%	-49.4%	-26.7%
Amphibians	-25.8%	?	-31.2%	?
Reptiles	-10.6%	?	-12.8%	?
Birds	-20.0%	-14.7%	-24.2%	-17.8%
Mammals	-22.7%	?	-27.4%	?

The results suggest that, very approximately, in 2015 the extent and quality of forest habitats was half of what would be present in the absence of bioenergy demand from the RED. For species the reduction in population size is about a quarter. These results may overestimate the degree to which this is driven by bioenergy demand, as some forestry activities (eg. thinning of stands) would occur to some extent anyway; and where economically viable, some biomass would be directed to other uses. Nevertheless, it is certain that bioenergy production in forests is of sufficient magnitude to lead to substantial habitat declines and population level impacts that are likely to affect the conservation status of many EU habitats and species.

The projections for future bioenergy feedstock used from forests (chapter 2) provide a high level of certainty that there will be continued growth in demand and production. Whilst it is very difficult to quantify this in terms of how it will increase the exposure of EU protected habitats and species to the risk factors, it is estimated that the bioenergy demand component for forest biomass will increase from 25% to approximately 30% (Figure 4-3). As indicated in the results, this will lead to further increases in the vulnerability of habitats and species, which will undoubtedly have greater potential to lead to substantial impacts on their conservation status.

Furthermore, it is also likely that the indirect effects of the demand from forest biomass are adding to the pressures on EU protected habitats and species. As indicated in Figure 4-11, a substantial proportion of habitats and species are considered to have medium or high vulnerability to ILUC. However, there is a very high coverage of forest habitats within the Natura 2000 network, with 77 of the 80 considered here having more than 75% coverage, one having 50 – 75% coverage, and the remaining two having missing data. Therefore, if the forests are managed appropriately within the network to reduce the impacts of the identified risk factors to acceptable levels, then the residual impacts would be considerably reduced to low levels, being no more than -0.7% (Figure 4-10). But this is not so much the case with HD and BD species, as their Natura 2000 coverage is more variable, with 28% having more than 75% coverage, and 16% having 50% - 75% coverage, but 32% had less than 25% coverage. Therefore their estimated levels of residual vulnerability remain relatively high, being approximately half of the estimates of potential vulnerability. However, coverage data were missing for 113 species (61%), and therefore the residual scores should be treated with some caution. Also it was not possible to separately estimate residual scores for amphibians, reptiles and mammals.

Figure 4-11 Estimates of the vulnerability of EU protected habitats and species to indirect land use change in forest areas as a result of biofuel feedstock demand

NB excludes habitats and species with zero general exposure in forest habitats

<u>HD forest habitats</u>

	ILUC vulnerability		
Group	**Low**	**Medium**	**High**
Boreal	13%	50%	38%
Temperate	8%	74%	18%
Mediterranean deciduous	15%	77%	8%
Mediterranean scleropyhllous	20%	80%	0%
Temperate mountain coniferous	67%	33%	0%

Mediterranean and mountain coniferous	50%	50%	0%
Overall	**18%**	**68%**	**13%**

BD and HD species that are predominantly associated with forest habitats

	ILUC vulnerability		
Group	**Low**	**Medium**	**High**
Plants	25%	33%	42%
Invertebrates	4%	96%	0%
Amphibians	0%	100%	0%
Reptiles	25%	75%	0%
Birds	40%	55%	5%
Mammals	7%	93%	0%
Overall	**15%**	**81%**	**4%**

5 Conclusions based on the bioenergy use review, sensitivity analysis and vulnerability assessment

5.1 The scope and limitations of the analysis.

This chapter aims to draw overall conclusions on the vulnerability of EU protected habitats and species to the production of bioenergy feedstocks in the EU. This is based on the results of the assessment of the 2015 baseline levels of bioenergy feedstock production in the EU and projections for 2030 (chapter 2), the review of the sensitivity of EU protected habitats and species to bioenergy feedstock production (chapter 3) and proceeding semi-quantitative analysis of the vulnerability of these habitats and species to 2015 and 2030 levels of bioenergy feedstock production. Before considering these conclusions, it is important to bear in mind the objectives, scope, and limitations of this aspect of the study, and the implications for its interpretation.

Firstly, this study has focused on EU protected habitats and species, i.e. those listed on Annexes I and II of the Habitats Directive, and Annex I of the Birds Directive, because they are of particular conservation concern in the EU, for example as a result of their rarity, uniqueness or fragility. Such habitats and species are therefore likely to be particularly sensitive to environmental changes, including those relating to bioenergy production. Furthermore, they either comprise natural / semi-natural habitats, or are species that mostly tend to be dependent on these types of habitats. Therefore, whilst this study's results are highly relevant to the EU's nature conservation policy priorities, they do not provide a representative indication of likely bioenergy feedstock production impacts on EU biodiversity in general (e.g. including common species that are found in intensively managed agricultural and forest areas).

It is also important to note that this study only considers biodiversity impacts in the EU, and does not consider those that may occur outside as a direct and indirect result of increasing bioenergy demand in the EU, although there is strong evidence that these may be substantial (Bowyer, 2010b; Meletiou et al, 2019).

Consideration of the results and conclusions from this study should also take into account the following key assumptions and constraints that affected the analysis that was undertaken. In particular:

- Whilst there is a good understanding of the ecological requirements of most EU protected habitats and species, which was sufficient to reliably identify the main bioenergy associated risk factors affecting them, direct evidence of their sensitivity to them is limited. This is partly because it is difficult to ascribe some key pressures (e.g. conversion of grassland to arable) to their drivers, including bioenergy feedstock demand. Also some bioenergy feedstocks (e.g. Miscathus), or agricultural / forestry practices (e.g. stump removal) are relatively new and/or uncommon, and have not been widely studied.
- Due to the limited direct evidence, assessment of the sensitivity of habitats and species to the risk factors was often based on inference from a knowledge of the habitats' and species' ecology, and the general effects of agricultural and forestry practices on them. Whilst there was sufficient information to reliably infer sensitivity for most risk factors for most habitats and species, there was more uncertainty over

some forestry practices (as further discussed below). Also whether some agricultural and forest practices have positive or negative impacts depends to a large extent on their scale and context, e.g. the scale of bioenergy crop planting and afforestation, clear cut area, the biodiversity value of the exiting effected habitat.

- The calculation of the 2015 baselines for the production of each type of bioenergy feedstock was constrained by challenges associated with being able to make connections between data sets on production of biomass material and their end use. This is because at the point of production of biomass material the end use for energy is often not clear, biomass production for energy is integrated with wider production for food, feed, materials etc. Clearly attributing volume change associated with energy and then linking this back to infield practices is challenging and leads to data gaps
- The development of the 2030 projections for the production of each type of bioenergy feedstock was hampered by a lack of clarity as to the exact level of bioenergy that would be utilised to 2030 and the associated consequences for land use. Modelling exercises have been run looking at changing demand for biomass based energy, however, only some look at land use consequences and often the detail of this at the level needed to understand impacts on protected habitats and species is absent.
- The extent to which habitats and species are actually exposed to each of the biofuel feedstock types was difficult to assess as the location of their production is often uncertain, or not directly ascertainable (e.g. for conventional crops that may be used for food, forage or bioenergy). Furthermore, it was necessary to consider the extent to which exposure may result from the conversion of habitats or low levels of use for feedstock production. Thus, for example, all HD forest types are considered to be potentially exposed to biomass production for bioenergy purposes to some extent (i.e. had a general exposure level of >5%).
- With some bioenergy feedstocks (e.g. SRC) it is evident that their presence is entirely, or almost entirely, driven by bioenergy demand. It is therefore relatively straightforward to relate the exposure of each habitat and species to their respective risk factors to bioenergy demand. However, it was much more difficult to assess the degree to which some forestry related risk factors are driven by bioenergy feedstock, because some sources of forest biomass, such as thinnings, may be removed to some extent for other forestry purposes (see further discussion below).
- It was not possible to take into account all the possible mitigation measures in the estimation of residual impacts. This was partly due to insufficient data being available on the effectiveness of such mitigation measures (see discussion inthe project's case studies), but also because many mitigation measures overlap and therefore their added value is unknown. Instead, residual impacts were only based on the assessment of Natura 2000 coverage (i.e. the proportion within the network) and the assumption that this would provide full protection for habitats and species within them. As discussed in chapter 2, there is evidence that protection levels in Natura 2000 are not complete (e.g. in relation to the ploughing up of seminatural habitats, and increasing forest management intensity). Therefore, in reality, the true vulnerability scores probably lie somewhere between the estimates of potential and residual vulnerability. Estimates of Natura 2000 coverage were missing for a high proportion of HD and BD species (over 61%), and therefore the estimates of their residual vulnerability scores should be treated as being indicative.

- The assessment of the vulnerability of habitats and species to bioenergy feedstock production was primarily based on their direct impacts, but other studies have shown that in the EU indirect land use change (ILUC) resulting from the increased demand for the feedstocks may lead to more widespread impacts (e.g. the intensification of management on seminatural habitats). As there is insufficient information available to assess such impacts reliably it was only possible to carry out a indicative assessment of the degree to which each EU protected habitat and species might be susceptible to ILUC. As this assessment was primarily based on expert judgement, the results should be treated with caution and considered to be only indicative.

Given these limitations, it is recommended that the individual habitat and species assessments, (available in a separate Excel file), should be considered to be indicative and not used a source of reference for predicted impacts. Instead, reference should be made to the combined assessments of the habitats and species as reported in chapter 3. Despite the data limitations and other constraints on the analysis, the results of this study provide a high level of certainty on the main bioenergy related risk factors for EU protected habitats and species, and the magnitude of impacts arising from the production of each of the main types of bioenergy feedstock in 2015 and as projected for 2030. These key results are set out in [Tables 5-1 to 5-3] and summarised below in relation to each of the types of bioenergy feedstock.

5.2 Conventional crops (i.e. food and feed/forage crops)

Evidence from the wider agricultural literature clearly shows that most EU protected habitats and species are highly sensitive to the risk factors associated with the production of bioenergy feedstocks from conventional crops (e.g. conversion to arable, increasing intensification of management). This is particularly the case where the growing of such crops leads to the conversion of seminatural grassland and other seminatural habitats (e.g. heathland, Mediterranean shrubland). Similar impacts occur where seminatural grassland management is intensified, such as to produce silage, but it is believed that this is rarely driven by a direct demand for bioenergy feedstocks.

In contrast, the use of existing arable land for bioenergy feedstocks has no impacts on HD habitats as they are absent. Similarly impacts on HD and BD species are much less than arise from the conversion of grassland etc, as typically only a few such species occur. These are mainly birds, and also tend to be restricted to areas of low intensity cereal production (such as in the dry steppe lands of Spain). Moreover, as the management of conventional crops is the same whether it is produced for food or for biofuels, the impacts are neutral if there are no changes in crop type. However, increasing demand for bioenergy feedstocks is known to be leading to changes in crop type in some areas. This is mainly leading to a switch from cereals to oilseed rape, which is unlikely to have significant impacts on EU protected species as few use such crops, and the ecological consequences from the change tend to be low and mixed. Bioenergy demand has also led to a considerable increase in the production of maize in some areas (on existing arable land and converted grassland) and this undoubtedly has detrimental impacts on a wide range of species (i.e. biodiversity in general). But impacts on EU protected species from the increase in maize on existing arable land are probably low, as few such species are present in conventional arable farmland. For similar reasons, whilst the removal of residues from croplands may have some detrimental impacts on biodiversity in general (e.g. from reductions in organic matter) it is unlikely to have any effect on EU protected species or habitats.

The analysis of the use of conventional crops for bioenergy feedstock production clearly shows that in the baseline year of 2015 a very small proportion of EU farmland was being used for this purpose, with only about 0.02% of grassland and cropland being converted to conventional crops for bioenergy feedstocks, and only about 0.5% affected by changes in crop type (Figure 4-3). Thus, overall exposure of EU protected habitats and species to the risk factors associated with the production of bioenergy through conventional crops is low. However, it should be borne in mind that the distribution of these crops is uneven, and that some areas have been subject to high levels of conversion (e.g. increases in oilseed rape crops in parts of south-east Europe) and large increases in maize production in Germany. In such areas impacts on EU protected habitats and species may be locally significant, especially where they have resulted from the conversion of grassland or other seminatural habitats to conventional crops.

As the projections for 2030 do not indicate a substantial change in the use of conventional crops for bioenergy feedstock, then no significant change in impacts are expected. Although a considerable increase is expected in the use of crop residues for bioenergy feedstocks (potentially affecting 7% of cropland) this is unlikely to affect any EU protected species, for the reasons described above.

Figure 5-1 Overall assessment of the vulnerability of EU protected habitats and species - Conventional crops (i.e. food and feed/forage crops)

Risk factor	Overall sensitivity of HD habitats	Overall sensitivity of HD and BD species	2015 baseline		2030 projection	
			Exposure level	Overall vulnerability	Exposure level	Overall vulnerability
Conversion of grassland or other semi-natural habitat or fallow to bioenergy feed crop	Very highly negative as all HD habitats are natural / semi-natural and therefore destroyed by conversion	Very highly negative: Vast majority of affected species lost	Very low: grassland conversion to arable crops is significant, but very small component for bioenergy (c. 0.02%)	Very low vulnerability due to very low exposure	No substantial change anticipated	Very low
Intensification of grassland management – e.g. higher fertiliser and pesticide use, re-seeding of grassland	Highly negative: All semi-natural habitats highly sensitive	Highly negative: Most species decline and some lost	No data available but known to be extremely low	Extremely low	No substantial change anticipated	Extremely low
Crop changes on existing arable land – cereals etc to oilseed rape and similar crops	None: as no HD habitats on arable land	Low: impacts are low and variable negative / positive and only affect a few EU protected species that	Low in most of the EU but moderate in some areas	Low,	No substantial change anticipated	Low

		use cereal crops (losses of ground nesting birds during harvest)				
Crop changes on existing arable land – cereals etc to maize monoculture	None: as no HD habitats on arable land	Highly negative, but only affect a few EU protected species that use cereal crop	Low overall, but some areas have high proportions of grassland and arable converted to maize crops (e.g. Germany)	Low overall, but locally high (e.g. Germany)	No substantial change anticipated	Low overall, but locally high (e.g. Germany)
Removal of residues from croplands	None: as no HD habitats on arable land	Unlikely to have significant impacts	Low (c. 0.8% of farmland)	Very low due to insignificant sensitivity	Substantial increase expected to about 7% of farmland	Very low despite increase due to low sensitivity of species

5.3 Bioenergy crops on agricultural land

Although relatively few studies have been carried out of the biodiversity impacts of dedicated bioenergy crops (e.g. Miscanthus and SRC), it is clear that they result in major changes in vegetation structure and composition. Thus, they effectively destroy any HD habitats that are present if they are grown on grassland or other seminatural habitats and have highly negative impacts on associated species. Similarly, afforestation of seminatural grassland and other seminatural habitats, leads to the loss of HD habitats and their associated species. In some circumstances appropriate small-scale afforestation using native trees can provide benefits for some species (e.g. nesting sites, ecological connectivity) in landscapes where trees have been removed. However, in most cases afforestation is not carried out for such conservation purposes. Where bioenergy crops and afforestation affect arable land then the impacts on EU protected habitats and species is likely to be much lower. This is primarily because HD habitats are absent on arable land and few EU protected species occur (as discussed above in relation to conventional bioenergy crops).

It is also clear from the available data that the extent of these bioenergy crops on farmland in the EU was extremely low in 2015, i.e. about 0.005% of farmland (Figure 4-3). Data on the extent to which afforestation is being carried out and driven by bioenergy demand is lacking, but it is also considered to affect a low proportion of grassland and seminatural habitat (c. 0.05%) and is minimal on cropland. There is a high level of certainty that the use of bioenergy crops will increase greatly (by several thousand percent) over the coming years in the EU (Figure 2-26). As a result, it is anticipated that approximately 1% of both grasslands and croplands will be affected by 2030. Similarly, afforestation on grasslands and other seminatural habitats is expected to increase tenfold, to affect about 0.5% of such habitats (but with no increase anticipated).

Despite this high-level of increase, impacts will probably remain low where bioenergy crops are grown on existing arable land. However, they will be more substantial where they lead to the loss of seminatural grasslands and other seminatural habitats. Furthermore, it is also important to bear in mind that it is anticipated that a high proportion of new areas of bioenergy crop will be located in marginal farming areas, e.g. in areas with poor soils, and therefore a high proportion will be on seminatural habitats. Consequently, whilst the overall exposure of farmland to these crops is relatively low (c 1%) EU protected habitats and species will be at a disproportionately higher risk of impact, with the potential for significant local losses unless areas with these habitats and species are avoided. In addition, the current and increasing demand for bioenergy feedstocks from conventional and bioenergy crops will also lead to ILUC impacts. Whilst these could not be quantified in this study, an expert evaluation of the potential vulnerability of EU protected habitats and species in agricultural areas was carried out, which suggests that a significant proportion have a medium or high vulnerability to habitat loss or degradation through ILUC (Figure 4-9).

The analysis presented in Figure 4-8 indicates the vulnerability of EU protected habitats and species to the combined risk factors associated with bioenergy feedstock production in agricultural areas, both in terms of potential vulnerability (i.e. with no mitigation measures) and residual vulnerability based on assumed full protection within the Natura 2000 network. This indicates that the coverage of most HD agricultural habitats within Natura 2000 network is relatively high. Therefore, if fully protected, the vulnerability of these habitats would drop to very low levels, even under the 2030 projections (from a mean potential vulnerability score of about -0.34% to a residual score of 0.027% with full Natura 2000 protection). The coverage of agricultural species within the Natura 2000 network is not so high, but full protection would still reduce residual impacts. As a result, if fully protected in the network the low potential vulnerability score for 2030 of -0.48% is estimated to fall to a residual level of approximately -0.33% (although the exact level is uncertain due to a high proportion of species having unknown Natura 2000 coverage). Other mitigation measures that could not be incorporated into the analysis (e.g. CAP cross compliance and greening measures) also have the potential to play an important role in reducing impacts if properly designed and implemented.

Figure 5-2 Overall assessment of the vulnerability of EU protected habitats and species to bioenergy crops on agricultural land

Risk factor	Overall sensitivity of HD habitats	Overall sensitivity of HD and BD species	2015 baseline		2030 projection	
			Exposure level	**Overall vulnerability**	**Exposure level**	**Overall vulnerability**
Conversion of arable cropland to Miscanthus or other non-food biomass crops or SRC	None: as no HD habitats on arable land	Low-level impacts which are likely to be variable depending on species involved for a relatively small number of cropland species (mainly birds)	Extremely low (c. 0.005% of agricultural land)	Extremely low due to minimal exposure and low sensitivity	Substantial increase expected to (c. 1.1%)	Despite the increase overall vulnerability will be relatively low
Conversion of grassland	Very highly negative:	Very highly negative - most	Extremely low (c.	Very low due to minimal	Substantial increase	Despite the high increase

or other semi-natural habitat to Miscanthus or other non-food biomass crops or SRC	conversion most likely of poor agricultural, ie semi-natural habitats, which will be destroyed	associated species of semi-natural habitats will be lost	0.005% of agricultural land – as above)	exposure despite high sensitivity	expected to (c. 1.3%)	overall vulnerability will be relatively low, but high local impacts possible, especially if expansion is mainly on semi-natural habitats
Afforestation of grassland or other semi-natural habitat	Very highly negative: afforestation most likely on poor agricultural, ie semi-natural habitats, which will be destroyed	Very highly negative - most associated species will be lost. Some biodiversity benefits possible in certain situations – eg if small patches link up fragmented habitats, or on improved grasslands	Very low (c. 0.05%	Low due to low exposure despite high sensitivity	Expected to increase ten-fold to c.0.5%	Low overall, but high local impacts depending on location, especially if expansion is mainly on semi-natural habitats
Afforestation of cropland	None: as no HD habitats on arable land	Very high negative impacts but on a small number of cropland species (mainly birds)	No data available but known to be extremely low	Extremely low	No change anticipated	Extremely low

5.4 Forest biomass from existing forest land

Forestry practices associated with the production of biomass for bioenergy feedstocks are similar to those undertaken for timber etc. Although the impact of such practices vary according to forest type and species, they show a general pattern of increasing detrimental impacts on biodiversity as the intensity of forest management increases. In fact, in some circumstances, low levels of forest management can be beneficial for some EU protected habitats and species, for example where selective logging or thinning opens up otherwise undermanaged forests with an even age structure and low structural diversity. Therefore, forest management for the low-level production of bioenergy feedstocks can sometimes be beneficial where this is compatible with biodiversity conservation objectives (e.g. as set for Natura 2000 sites). In fact, the revenue created from biomass for bioenergy may help to support forest management practices, such as coppicing, which is required for the conservation of some EU protected habitats and species.

More often forest biomass production for bioenergy will entail more intensive and large-scale forestry management which is detrimental for biodiversity, especially EU protected habitats and species. In this respect the removal of deadwood is highly detrimental, whilst the clearcutting of forests and their replanting is even more damaging, as it effectively destroys HD habitats and results in the losses of the majority of associated species. Whilst small-scale clearcutting may be less damaging, especially where mitigated through the retention of some dead and live trees, and may even be beneficial in the short term for some species, it is more often damaging for EU protected species. Therefore, whilst there is some uncertainty over the sensitivity of some species to some forestry practices (e.g. more recent and less common practices such as stump removal), it is clear that the overall impacts of typical forestry practices are damaging for the majority of EU habitats and species, whether they are carried out to provide bioenergy or other forest products.

The review in chapter 2 of information on the use of forest products estimates that in 2015 approximately 25% of forest biomass was being used for bioenergy feedstock purposes (Figure 2-25). Therefore, whilst it is difficult to estimate the proportion of EU protected forest habitats and species that occur in areas where forest biomass is being taken for bioenergy feedstocks, it is clear that a substantial proportion are generally exposed to the associated forestry management practices and their associated biodiversity risk factors. It is, however, much more difficult to estimate the exposure of the habitats and species to the individual risk factors associated with the biomass production. This is primarily because information on the extent to which each of the forestry practices is carried out in forest types that holds EU protected habitats and species is lacking. Furthermore, some practices are fairly recent and appear to be increasing (e.g. stump removal and whole tree harvesting).

It must also be borne in mind that some forestry practices (e.g. thinning and stump removal) would be carried out to some extent even in the absence of bioenergy demand, such as to increase timber production. No data could be found to quantify the extent to which these practices occur in the absence of bioenergy demand (i.e. to establish the counterfactual situation). The vulnerability analysis in chapter 3 assumed that these practices would not continue in the absence of bioenergy demand. This therefore probably results in an overestimate to some extent of the bioenergy driven impacts in forests. Nevertheless, although the quantification is difficult there is reasonable certainty that several of the more damaging forestry practices are being commonly carried out, and that the demand for biology feedstocks is a significant driver of these practices.

Taking into account the sensitivity of EU protected habitats and species to forestry practices and their exposure to them, there is a high degree of certainty that they are highly vulnerable to the removal of deadwood, clear cutting of trees and re-planting, and to a lesser degree other forestry risk factors, resulting from the demand for bioenergy feedstocks in 2015. As indicated in Figure 4-10, this gives rise to a very high mean potential vulnerability score for 2015 of -46% for HD habitats and -22% for HD and BD species. This almost certainly means that the conservation status of the affected habitats and species is likely to be compromised in the areas where forestry management is being carried out intensively. Furthermore, it is widely anticipated that the demand for biomass from forests for bioenergy will increase, such that by 2030 it is estimated that about 30% of forest biomass will be used for such purposes **Error! Reference source not found.**. Based on this projection the mean vulnerability score will i ncrease to approximately -57% for habitats and -27% for species.

It is also necessary to consider additional indirect land-use changes that may occur from the current and projected very high levels of demand for biomass for bioenergy feedstocks from forests. Whilst this could not be quantified in this study, expert assessments suggest that a high proportion of EU protected habitats and species of forests have at least a moderate level of vulnerability to ILUC (Figure 4-11).

Given the current and projected very high vulnerability of EU protected habitats and species to the production of forest biomass for bioenergy, it is especially important to consider the degree to which the potential impacts may be avoided through protection within the Natura 2000 network. Figure 4-10 indicates that, because the coverage of HD forest habitats within Natura 2000 network is very high, the residual vulnerability of these habitats would be low (-0.49% in 2015 and -0.59% in 2030). Although the coverage of forest species within the Natura 2000 network is not as high, full protection would still reduce residual impacts to about half (-13%. for 2015 and -15% for 2030). However, as was the case with agricultural species, the exact level is uncertain due to a high proportion of species having unknown Natura 2000 coverage. It should also be borne in mind that the residual impact estimates assume full protection, which as discussed above, is not realistic. On the other hand, other mitigation measures are not taken into account, such as forest certification schemes or forest-environment contracts (see case studies in the main report). The true vulnerability scores probably therefore lie between the potential and the residual estimates. Irrespective of this uncertainty, it is obvious that the protection of the Natura 2000 network has a crucial role to play in reducing the effects of bioenergy feedstock production that are undoubtedly having substantial impacts of on EU protected forest habitats and species.

Figure 5-3 Overall assessment of the vulnerability of EU protected habitats and species to removal of forest biomass from existing forest land

Risk factor	**Overall sensitivity of HD habitats**	**Overall sensitivity of HD and BD species**	**2015 baseline**		**2030 projection**	
			Exposure level	**Overall vulnerability**	**Exposure level**	**Overall vulnerability**
Selective logging of trees in extensively managed forests	Moderately to highly negative depending on % extracted, methods and forest age and condition; unless required to meet conservation objectives	Moderately negative for most species depending on % extracted, methods and forest age and condition; unless required to meet conservation objectives	Moderate: c. 2.5% forest affected as a result of bioenergy demand	Variable depending on species and methods, but generally moderate	Expected to increase slightly to about 3% exposure	Small increase, but remaining generally moderate
Thinning of small and intermediate sized successional trees	Low and mostly negative, but beneficial in undermanaged non-natural forest	Low and mostly negative for a few species, but beneficial in undermanaged non-natural forest (detrimental if carried out in bird breeding season)	High: common practice, c. 19% driven by bioenergy demand	Variable depending on species and forest condition, but generally moderately negative	Expected to increase slightly	Small increase, remaining generally moderately negative

Risk factor	Overall sensitivity of HD habitats	Overall sensitivity of HD and BD species	2015 baseline		2030 projection	
			Exposure level	Overall vulnerability	Exposure level	Overall vulnerability
Removal of dead wood	Highly negative	Highly negative, especially for a large proportion of mosses and liverworts, invertebrates and specialist forest birds	High: common practice, c. 16% driven by bioenergy demand	High, due to exposure level and high sensitivity for habitats and many species	Expected to increase slightly	High
Clear cutting of trees to increase production efficiency	Very highly negative – as leads to habitat destruction, and very slow recovery	Highly negative for most species, especially if at large scale; creates temporary habitat for a few species	High: c. 16% of forest affected as a result of bioenergy demand	High, due to exposure level and high sensitivity for habitats and many species	Expected to increase slightly	High
Stump removal and whole-tree harvesting	Very highly negative – as leads to major habitat degradation	Information lacking, but probably highly negative, especially for a large proportion of mosses and liverworts, invertebrates	Uncertain, but common practice in some regions (e.g. Sweden and Finland) - overall exposure estimated to be very approximately 3%	Low overall but locally high for some species (e.g. invertebrates)	Expected to increase	Uncertain, but locally high
Tree planting and use of pesticides and fertilisers	Very highly negative – as leads to habitat destruction, and very slow recovery	Highly negative for most species	High: tree planting is a common practice, c. 17% driven by bioenergy demand (but exposure to fertiliser and pesticide use less certain)	Moderate to high depending on species	Expected to increase slightly	Small increase, remaining generally moderate to high
Plantation forestry with dominance of non-native or non-site-typical tree species	Very highly negative – as leads to habitat destruction	Highly negative for almost all species	Low: c. 0.3% of forest affected as a result of bioenergy demand	Moderate, due to very high sensitivity	Expected to increase slightly	Moderate
Construction of forest roads and tracks to support forest managemen t and harvesting.	Low from footprint, but moderate impacts can arise from hydrological disruption for some habitat types. Paved	Roads create fragmentation of populations unable to traverse or lead to road mortality of some species. Unpaved low	No data, and difficult to estimate but considered to be about 1.25% direct exposure	Low to moderate depending on the species	Expected to increase slightly	Low to moderate

Risk factor	Overall sensitivity of HD habitats	Overall sensitivity of HD and BD species	2015 baseline		2030 projection	
			Exposure level	Overall vulnerability	Exposure level	Overall vulnerability
	roads with traffic fragment habitats.	traffic tracks have no impacts for most species, but low to moderate disturbance impacts for some				

5.5 Overall conclusions

The clearest and most important conclusion from this assessment of the vulnerability of EU protected habitats and species to bioenergy feedstock production, is that the use of forest biomass (round wood and residues etc) is almost certainly already having a significant impact on most HD forest habitats and many forest species (through its direct impacts as well as indirect land use change). This is because such habitats and species are sensitive to a number of common intensive forestry practices and a significant driver of this is the demand for forest biomass, as approximately 25% of biomass that is extracted from forests is used for this purpose. It is also widely expected that the demand for forest biomass for bioenergy will increase, with this study projecting that by 2030 30% of extracted forest biomass will be for bioenergy use.

In contrast, the impacts of bioenergy feedstock production from dedicated bioenergy crops (e.g. Miscathus and SRC) or trees planted on farmland, on EU protected habitats and species are currently very low, primarily because very small areas are affected. Furthermore, the afforestation of arable farmland and intensively managed grasslands, or their conversion to bioenergy crops, has relatively low impacts on EU protected habitats and species. This is mainly because HD habitats are absent and few HD and BD species occur in significant amounts on farmland, although there are some areas with low intensity cereal production that are important for some species, mainly birds. However, the afforestation or conversion of seminatural grasslands, and other seminatural habitats, to bioenergy crops has the potential to lead to considerable impacts as a high proportion of such habitats and their associated species are the focus of protection under the Nature Directives. Furthermore, afforestation or the planting of bioenergy crops destroys the HD habitats and leads to the loss of most associated species. The impacts of afforestation and bioenergy crops may therefore be much more significant in future as it is anticipated that a substantial expansion of the production of these crops will occur. Whilst the expansion is currently expected to only affect about 1% of farmland, it is expected that marginal areas of farmland will be mainly targeted, and therefore areas of seminatural habitat will be at a disproportionately higher level of risk. This may give rise to significant impacts on some EU habitats and species in some areas.

The use of conventional crops (e.g. oil-seed rape and maize) for bioenergy is also currently considered to be having low impacts on EU protected habitat and species. This is also partly because such habitats and species are largely absent from arable farmland, but also because the proportion of these crops being used for bioenergy is currently low. Furthermore, the crops are principally managed in the same way whether they are used for bioenergy or food etc, and the impacts of changes in crop type are generally minimal. The only exception to this that significantly affects EU protected species, is with respect to maize. Some areas of Europe (e.g. Germany) have seen large increases in the growing of maize, in part for bioenergy uses. This has led to the conversion of grasslands for this purpose as well as increases in crop management intensity compared to most arable crops, and this is associated with noticeable negative effects on some BHD species groups such as amphibians. At the moment no major changes are anticipated in the use of conventional crops for bioenergy purposes up to 2030, so impacts are unlikely to increase, but other drivers (such as changes in the dairy sector) are still contributing to an increasing maize are.

The study has also highlighted the major role that the Natura 2000 network has to play in the protection of EU protected habitats and species from the potential impacts of bioenergy feedstock production. This is because a very high proportion of the area of most types of HD agricultural and forest habitat occur within the network, and, probably, a substantial proportion of most HD and BD species. Thus, the proper and full implementation of the Habitats Directive's Natura 2000 site protection and management measures, in combination with other policy measures (e.g. CAP regulations and funding), should be sufficient to avoid many of the potential impacts. In this respect it is important to emphasise that this does not necessarily require that bioenergy feedstock production is prohibited within Natura sites – but instead that it should be compatible with the site conservation objectives for its respective habitats and species.

However, EU protected habitats and species also occur outside the Natura 2000 network to some extent, and may therefore be at high risk of impacts from bioenergy production, particularly in forests, and in semi-natural habitats that may be targeted for afforestation or conversion to bioenergy crops. Other mitigation measures will therefore be needed to avoid and reduce impacts to acceptable levels in these areas, in particular agreements and criteria for biomass from nature conservation management (inside or outside Natura 2000), or forest certification and sustainable management or forest-environment contracts (see case studies).

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Annex 1. HD and BD species associated with agricultural, heath/scrub and forest habitats included in the scope of this study

HD and BD species associated with CROPLAND ecosystem (arable, permanent crops, and farmland mosaics) as preferred habitat

Species group	**HD and BD species associated with CROPLAND ecosystem (farmland mosaics with arable) as preferred habitat**
birds (birds of prey and owls)	Levant Sparrowhawk (*Accipiter brevipes*) Cinereous Vulture (*Aegypius monachus*) Short-eared Owl (*Asio flammeus*) Montagu's Harrier (*Circus pygargus*) Red-footed Falcon (*Falco vespertinus*) Lesser Kestrel (*Falco naumanni*)
birds (insectivores and mixed diet)	Greater Short-toed Lark (*Calandrella brachydactyla*), Calandra Lark (*Melanocorypha calandra*) Corncrake (*Crex crex*) Eurasian Golden-plover (*Pluvialis apricaria*) European Roller (*Coracias garrulous*) Syrian Woodpecker (*Dendrocopos syriacus*) Ortolan Bunting (*Emberiza hortulana*) Olive-tree Warbler (*Hippolais olivetorum*) Red-backed Shrike (*Lanius collurio*) Lesser Grey Shrike (*Lanius minor*) Masked Shrike (*Lanius nubicus*) Common Crane (*Grus grus*) Iberian Grey Partridge (*Perdix perdix hispaniensis*) Great Bustard (*Otis tarda*) Little Bustard (*Tetrax tetrax tetrax*)
birds (wintering waterfowl)	Greenland White-fronted Goose (*Anser albifrons flavirostris*), Lesser White-fronted Goose (*Anser erythropus*), Barnacle Goose (*Branta leucopsis*) Tundra Swan (*Cygnus columbianus bewickii*), Whooper Swan (*Cygnus cygnus*)
mammals (carnivores)	Steppe Polecat (*Mustela eversmanii*)
mammals (rodents)	European Hamster (*Cricetus cricetus*), Souslik (*Spermophilus citellus*), Spotted Souslik *(Spermophilus suslicus*), Severtzov's Birch Mouse (*Sicista subtilis*)
mammals (bats)	*Miniopterus schreibersii, Eptesicus nilssonii, Myotis emarginatus, Myotis myotis, Myotis punicus, Pipistrellus kuhlii, Plecotus austriacus, Rhinolophus ferrumequinum, Rhinolophus hipposideros, Rhinolophus mehelyi*
reptiles	Maltese Wall Lizard (*Podarcis filfolensis*) Milos Wall Lizard (*Podarcis milensis*)
amphibians	Green Toad (*Bufotes viridis*) Common Spadefoot Toad (*Pelobates fuscus*)
arthropods	Balkan Pincer Grasshopper (*Paracaloptenus caloptenoides*), longhorn beetle (*Pilemia tigrina*), Eastern Eggar moth (*Eriogaster catax*), Raetzer's Ringlet (*Erebia christi*), Danube Clouded Yellow (*Colias myrmidone*)
plants	*Notothylas orbicularis, Bromus grossus, Linaria ricardoi, Agrimonia pilosa*

HD and BD species associated with GRASSLAND (pastures, meadows and natural grasslands) and/or HEATH/SCRUB ecosystem as preferred habitat

Species group	**HD and BD species associated with GRASSLAND and/or HEATH/SCRUB ecosystems as preferred habitat**
birds (birds of prey and owls)	Golden Eagle (*Aquila chrysaetos*), Imperial Eagle (*Aquila heliaca*), Spanish Imperial Eagle (*Aquila adalberti*), Lesser Spotted Eagle (*Aquila pomarina*) Long-legged Buzzard (*Buteo rufinus*) Short-toed Snake Eagle (*Circaetus gallicus*) Northern Hen Harrier (*Circus cyaneus*), Pallid Harrier (*Circus macrourus*), Montagu's Harrier (*Circus pygargus*) Black-winged Kite (*Elanus caeruleus*) Saker Falcon (*Falco cherrug*), Lanner Falcon (*Falco biarmicus*), Gyr Falcon (*Falco rusticolus*), Lesser Kestrel (*Falco naumanni*), Merlin (*Falco columbarius*) Short-eared Owl (*Asio flammeus*), Snowy Owl (*Bubo scandiaca*)
birds (insectivores and mixed diet)	Greater Short-toed Lark (*Calandrella brachydactyla*), Calandra Lark (*Melanocorypha calandra*), Thekla Lark (*Galerida theklae*), Dupont's Lark (*Chersophilus duponti*), Wood Lark (*Lullula arborea*) Tawny Pipit (*Anthus campestris*) Nightjar (*Caprimulgus europaeus*) Stone Curlew (*Burhinus oedicnemus*) Eurasian Golden-plover (*Pluvialis apricaria*) Collared Pratincole (*Glareola pratincola pratincola*) European Roller (*Coracias garrulous*) Ortolan Bunting (*Emberiza hortulana*) Red-backed Shrike (*Lanius collurio*) Trumpeter Finch (*Bucanetes githagineus*) Cyprus Wheatear (*Oenanthe cypriaca*), Canary Islands Stonechat (*Saxicola dacotiae*) Bluethroat (*Luscinia svecica svecica*) Aquatic Warbler (*Acrocephalus paludicola*) Dartford Warbler (*Sylvia undata* with *dartfordiensis*), Marmora's Warbler (*Sylvia sarda*), Rueppell's Warbler (*Sylvia rueppelli*), Barred Warbler (*Sylvia nisoria*) White Stork (*Ciconia ciconia ciconia*) Iberian Grey Partridge (*Perdix perdix hispaniensis*), Italian Grey Partridge (*Perdix perdix italica),* Barbary Partridge *(Alectoris barbara),* Rock Partridge *(Alectoris graeca graeca, whitakeri* and *saxatilis),* Pyrenaean Rock Partridge *(Lagopus muta pyrenaica)* Black Grouse (*Tetrao tetrix tetrix*), Great Bustard (*Otis tarda*), Little Bustard (*Tetrax tetrax tetrax*), Houbara Bustard (*Chlamydotis undulata*), Common Buttonquail (*Turnix sylvaticus*) Pin-tailed Sandgrouse (*Pterocles alchata*) and Black-bellied Sandgrouse (*Pterocles orientalis*)
birds (wintering waterfowl)	Greenland White-fronted Goose (*Anser albifrons flavirostris*), Lesser White-fronted Goose (*Anser erythropus*), Barnacle Goose (*Branta leucopsis*) Tundra Swan (*Cygnus columbianus bewickii*), Whooper Swan (*Cygnus cygnus*)
mammals (carnivores)	Grey Wolf (*Canis lupus*), Arctic Fox (*Alopex lagopus*), Eurasian Lynx (*Lynx lynx*), Iberian Lynx (*Lynx pardinus*), Wolverine (*Gulo gulo*), Wildcat (*Felis silvestris*), Steppe Polecat (*Mustela eversmanii*), Marbled Polecat (*Vormela peregusna*), Brown Bear (*Ursus arctos*), Golden Jackal (*Canis aureus*), Genette (*Genetta genetta*), Egyptian Mongoose (*Herpestes ichneumon*), European Polecat (*Mustela putorius*)
mammals (large grazers)	European Bison / Wisent (*Bison bonasus*), Spanish Ibex (*Capra pyrenaica pyrenaica*), Appenine Chamois (*Rupicapra pyrenaica ornata)*, Balkan Chamois (*Rupicapra rupicapra balcanica*), Tatra Chamois (*Rupicapra rupicapra tatrica*), Corsican Red Deer (*Cervus elaphus corsicanus*)
mammals (rodents & others)	European Hamster (*Cricetus cricetus*), Souslik (*Spermophilus citellus*), Spotted Souslik *(Spermophilus suslicus*), Severtzov's Birch Mouse (*Sicista subtilis*), Northern Birch Mouse (*Sicista betulina*), Romanian Hamster (*Mesocricetus newtoni*), Cabrera's Vole (*Microtus cabrerae*), Tundra Vole – Netherlands subspecies (*Microtus oeconomus arenicola*), Central European Tundra Vole (*Microtus oeconomus mehelyi*), Tatra Alpine Marmot (*Marmota*

	marmota latirostris), Sicilian Shrew (*Crocidura sicula*), Roach's Mouse-Tailed Dormouse (*Myomimus roachi*), Hazel Dormouse (*Muscardinus avellanarius*), Forest Dormouse (*Dryomys nitedula*), Crested Porcupine (*Hystrix cristata*) North African Hedgehog (*Erinaceus algirus*) Mountain Hare (*Lepus timidus*)
mammals (bats)	*Barbastella barbastellus, Eptesicus serotinus, Eptesicus bottae, Eptesicus nilssonii, Miniopterus schreibersii, Myotis alcanthoe, Myotis aurascens, Myotis emarginatus, Myotis blythii, Myotis bechsteinii, Myotis capaccinii, Pipistrellus pygmaeus, Plecotus austriacus, Plecotus kolombatovici, Plecotus macrobullaris, Rhinolophus ferrumequinum, Rhinolophus blasii, Rhinolophus hipposideros, Tadarida teniotis*
reptiles	Caspian Whip Snake (*Coluber caspius*), Cyprus Whip Snake (*Coluber cypriensis*), Large Whip Snake (*Coluber jugularis*), Dahl's Whip Snake (*Coluber najadum*), Leaden-Colored Racer (*Coluber nummifer*), Western Whip Snake (*Coluber viridiflavus*), Smooth Snake (*Coronella austriaca*), Aesculapian Snake (*Elaphe longissima*), Italian Aesculapian Snake (*Elaphe lineata*), Four-lined Snake (*Elaphe quatuorlineata*), Blotched Snake (*Elaphe sauromates*), European Ratsnake (*Elaphe situla*), Javelin Sand Boa (*Eryx jaculus*), Horseshoe Whip Snake (*Hemorrhois hippocrepis*), Cyclades Blunt-nosed Viper / Milos Viper (*Macrovipera schweizeri*), Soosan Snake (*Telescopus fallax*), Orsini's Viper / Meadow Viper (*Vipera ursinii* with *macrops* and *rakosiensis*), Ottoman Viper (*Vipera xanthina*), Seoane's Viper (*Vipera seoanei*) East Canary Gecko (*Tarentola angustimentalis*), Tenerife Wall Gecko (*Tarentola delalandii*) European Glass Lizard (*Pseudopus apodus*) Pyrenean Rock Lizard (*Iberolacerta bonnali*), Schreiber's Green Lizard (*Lacerta schreiberi*), Balkan Green Lizard (*Lacerta trilineata*), Sand Lizard (*Lacerta agilis*), Western Green Lizard (*Lacerta bilineata*), European Green Lizard (*Lacerta viridis*), Snake-eyed Lacertid (*Ophisops elegans*), Maltese Wall Lizard (*Podarcis filfolensis*), Tyrrhenian Wall Lizard (*Podarcis tiliguerta*), Sicilian Wall Lizard (*Podarcis wagleriana*), Erhard's Wall Lizard (*Podarcis erhardii*), Dalmatian Wall Lizard (*Podarcis melisellensis*), Common Wall Lizard (*Podarcis muralis*), Balkan Wall Lizard (*Podarcis tauricus*), Viviparous Lizard (*Zootoca vivipara ssp pannonica*), (*Podarcis lilfordi*), (*Podarcis cretensis*), (*Podarcis levendis*), (*Podarcis milensis*), (*Algyroides marchi*), (*Algyroides moreoticus*), (*Algyroides nigropunctatus*) Tenerife Lizard (*Gallotia galloti*), Giant Canary Island Lizard (*Gallotia stehlini*), Atlantic Lizard (*Gallotia atlantica*), Gallot's Lizard (*Gallotia galloti insulanagae)* European Copper Skink (*Ablepharus kitaibelii*), Bedriaga's Skink (*Chalcides bedriagai*), Gran Canaria Skink (*Chalcides sexlineatus*), West Canary Skink (*Chalcides viridanus*), Ocellated Skink (*Chalcides ocellatus*), Canarian Cylindrical Skink (*Chalcides simonyi*) Spur-thighed Tortoise (*Testudo graeca*), Hermann's Tortoise (*Testudo hermanni*), Marginated Tortoise (*Testudo marginata*), Sicilian Pond Turtle (*Emys trinacris*)
amphibians	Common Spadefoot Toad with Po subspecies (*Pelobates fuscus* and *insubricus*), Spanish Spadefoot Toad (*Pelobates cultripes*), Eastern Spadefoot Toad (*Pelobates syriacus*), Natterjack Toad (*Epidalea calamita*), Green Toad (*Pseudoepidalea viridis*), Common Midwife Toad (*Alytes obstetricans*), Iberian Midwife Toad (*Alytes cisternasii*), Painted Frog (*Discoglossus pictus*), Middle East Tree Frog (*Hyla savignyi*) Alpine Salamander (*Salamandra atra*), Golden Alpine Salamander (*Salamandra atra aurorae*), Salamandra di Lanza (*Salamandra lanzai*), Luschan's Salamander (*Mertensiella luschani)*
arthropods	Spider - *Macrothele calpeiana* Beetles (Coleoptera) - *Carabus hungaricus, Carabus zawadszkii, Dorcadion fulvum cervae, Bolbelasmus unicornis, Probaticus subrugosus, Pilemia tigrina, Carabus olympiae, Carabus variolosus, Pseudogaurotina excellens* Butterflies and moths (Lepidoptera) - *Eriogaster catax, Paracossulus thrips, Chondrosoma fiduciarium, Lignyoptera fumidaria, Phyllometra culminaria, Glyphipterix loricatella, Cucullia mixta, Gortyna borelii lunata, Polymixis rufocincta isolata, Lycaena dispar, Lycaena helle, Maculinea arion, Maculinea nausithous, Maculinea teleius, Plebicula golgus, Polyommatus eroides, Pseudophilotes bavius, Agriades glandon aquilo, Coenonympha hero, Coenonympha oedippus, Erebia calcaria, Erebia christi, Erebia medusa polaris, Erebia*

	sudetica, Melanargia arge, Proterebia afra dalmata, Colias myrmidone, Euphydryas aurinia Hesperia comma catena, Parnassius apollo, Parnassius mnemosyne, Zerynthia polyxena, Papilio alexanor, Papilio hospiton, Fabriciana niobe elisa, Lopinga achine, Clossiana improba, Hyles hippophaes, Proserpinus proserpina, Callimorpha (Euplagia) quadripunctaria, Apatura metis, Erannis ankeraria Grasshoppers (Orthoptera) - *Odontopodisma rubripes, Paracaloptenus caloptenoides, Stenobothrus eurasius, Baetica ustulata, Isophya costata, Isophya harzi, Isophya stysi, Pholidoptera transsylvanica, Saga pedo, Apteromantis aptera*
molluscs	*Caseolus calculus, Caseolus commixta Helicopsis striata austriaca Idiomela subplicata, Discula turricula Hystricella leacockiana, Vertigo angustior, Vertigo geyeri, Vertigo moulinsiana, Lampedusa imitatrix, Discus guerinianus, Geomitra moniziana, Helix pomatia* Kerry Slug *(Geomalacus maculosus)*
plants	281 vascular plant spp

HD and BD species associated with FOREST ecosystem (forests, woodland, wooded pastures) as preferred habitat

Species group	HD and BD species associated with FOREST ecosystem as preferred habitat
birds (birds of prey, owls)	Cinereous Vulture (*Aegypius monachus*) Bonelli's Eagle (*Aquila fasciatus*), Lesser Spotted Eagle (*Aquila pomarina*), Short-toed Snake-eagle (*Circaetus gallicus*), White-tailed Sea-eagle (*Haliaeetus albicilla*), European Honey-buzzard (*Pernis apivorus*), Booted Eagle (*Hieraaetus pennatus*), Greater Spotted Eagle (*Aquila clanga*), Imperial Eagle (*Aquila heliaca*), Osprey (*Pandion haliaetus*), Levant Sparrowhawk (*Accipiter brevipes*), Black Kite (*Milvus migrans),* Red Kite (*Milvus milvus*), Red-footed Falcon (*Falco vespertinus*), Boreal Owl (*Aegolius funereus*), Earasian Eagle Owl (*Bubo bubo*), Eurasian Pygmy Owl (*Glaucidium passerinum*), Great Grey Owl (*Strix nebulosa*), Ural Owl (*Strix uralensis*), Northern Hawk Owl (*Surnia ulula*)
birds (insectivores and mixed diet)	La Palma Chaffinch (*Fringilla coelebs ombriosa*), Blue Chaffinch (Gran Canaria) (*Fringilla teydea polatzek*i), Wood Lark (*Lullula arborea*) Olive-tree Warbler (*Hippolais olivetorum*) Black Stork (*Ciconia nigra*) Nightjar (*Caprimulgus europaeus*) Dark-tailed Laurel-pigeon (*Columba bollii*), White-tailed Laurel-pigeon (*Columba junoniae)*, Azores Wood Pigeon (*Columba palumbus azorica*), Trocaz Pigeon (*Columba trocaz*) Scottish Crossbill (*Loxia scotica*) European Roller (*Coracias garrulous*), Red-backed Shrike (*Lanius collurio*) Collared Flycatcher (*Ficedula albicollis*), Red-breasted Flycatcher (*Ficedula parva*), Semicollared Flycatcher (*Ficedula semitorquata*), Krueper's Nuthatch (*Sitta krueperi),* Corsican Nuthatch (*Sitta whiteheadi*), White-backed Woodpecker (*Dendrocopos leucotos*), Middle Spotted Woodpecker (*Dendrocopos medius*), Syrian Woodpecker (*Dendrocopos syriacus*), Black Woodpecker (*Dryocopos martius*), Eurasian Three-toed Woodpecker (*Picoides tridactylus*), Grey-headed Woodpecker (*Picus canus*) Hazel Grouse (Bonasia bonasia), Black Grouse (continental) (Tetrao tetrix tetrix) Western Capercaillie (Tetrao urogallus with aquitanicus), Cantabrian Capercaillie (Tetrao urogallus cantabricus)
mammals (carnivores)	Grey Wolf (*Canis lupus*), Eurasian Lynx (*Lynx lynx*), Iberian Lynx (*Lynx pardinus*), Wolverine (*Gulo gulo*), Wildcat (*Felis silvestris*), Brown Bear (*Ursus arctos*)

mammals (large grazers)	European Bison / Wisent (*Bison bonasus*), Wild Sheep (*Ovis aries*), Finnish Forest Reindeer (wild) (*Rangifer tarandus fennicus*), Corsican Red Deer (*Cervus elaphus corsicanus*)
mammals (rodents & others)	European Beaver (*Castor fiber*), Tatra Pine Vole (*Microtus tatricus*), Northern Birch Mouse (*Sicista betulina*), Roach's Mouse-Tailed Dormouse (*Myomimus roachi*), Hazel Dormouse (*Muscardinus avellanarius*), Forest Dormouse (*Dryomys nitedula*), Siberian Flying Squirrel (*Pteromys volans*), Caucasian Squirrel (*Sciurus anomalus*) North African Hedgehog (*Erinaceus algirus*)
mammals (bats)	*Miniopterus schreibersii, Tadarida teniotis, Rousettus aegyptiacus, Rhinolophus blasii, Rhinolophus euryale, Rhinolophus ferrumequinum, Rhinolophus hipposideros, Rhinolophus mehelyi, Barbastella barbastellus, Eptesicus nilssonii, Eptesicus serotinus, Hypsugo savii, Myotis alcathoe, Myotis aurascens, Myotis bechsteinii, Myotis brandtii, Myotis capaccinii, Myotis dasycneme, Myotis daubentonii, Myotis emarginatus, Myotis escalerai, Myotis myotis, Myotis mystacinus, Myotis nattereri, Nyctalus azoreum, Nyctalus lasiopterus, Nyctalus leisleri, Nyctalus noctula, Pipistrellus maderensis, Pipistrellus nathusii, Pipistrellus pipistrellus, Pipistrellus pygmaeus, Plecotus auritus, Plecotus sardus, Plecotus teneriffae, Vespertilio murinus*
reptiles	Aesculapian Snake (*Elaphe longissima*), Italian Aesculapian Snake (*Elaphe lineata*), Blotched Snake (*Elaphe sauromates*) Viviparous Lizard (*Zootoca vivipara ssp pannonica*), Danford's Lizard *(Lacerta danfordi),* Oertzen's Rock Lizard (*Lacerta oertzeni*), Spanish Algyroides (*Algyroides marchi*), Greek Algyroides (*Algyroides moreoticus*), Dalmatian Algyroides (*Algyroides nigropunctatus*), Fitzinger's Algyroides (*Algyroides fitzingeri*) Gran Canaria Skink (*Chalcides sexlineatus*), West Canary Skink (*Chalcides viridanus*) Hermann's Tortoise (*Testudo hermanni*)
amphibians	Fire-bellied Toad (*Bombina bombina*), Yellow-bellied Toad (*Bombina variegata*), Common Midwife Toad (*Alytes obstetricans*), Middle East Tree Frog (*Hyla savignyi*), Moor Frog / Altai Brown Frog (*Rana arvalis*), Agile Frog (*Rana dalmatina*), Greek Stream Frog (*Rana graeca*), Italian Agile Frog (*Rana latastei*), Pool Frog (*Rana lessonae*) Golden-striped Salamander (*Chioglossa lusitanica*), Alpine Salamander (*Salamandra atra*), Golden Alpine Salamander (*Salamandra atra aurorae*), Salamandra di Lanza (*Salamandra lanzai*), Luschan's Salamander (*Mertensiella luschani)),* Spectacled Salamander *(Salamandrina terdigitata)* *Italian Crested Newt (Triturus carnifex), Northern/Great Crested Newt (Triturus cristatus), Danube Crested Newt (Triturus dobrogicus), Italian Newt (Triturus italicus), Balkan Crested Newt (Triturus karelinii), Macedonian Crested Newt (Triturus macedonicus), Marbled Newt (Triturus marmoratus), Carpathian Newt (Triturus montandoni), Transylvanian Smooth Newt (Triturus vulgaris ampelensis)*
arthropods	Arachnida- *Anthrenochernes stellae*, *Macrothele calpeiana* Hemiptera - *Aradus angularis* Beetles (Coleoptera) - *Xyletinus tremulicola, Boros schneideri, Stephanopachys linearis, Stephanopachys substriatus , Buprestis splendens,Carabus hampei, Carabus menetriesi pacholei, Carabus nodulosus, Carabus olympiae, Carabus variolosus,Carabus zawadszkii, Corticaria planula,Cerambyx cerdo, Mesosa myops, Morimus funereus, Pseudogaurotina excellens, Rosalia alpina, Osmoderma eremita, Cucujus cinnaberinus, Limoniscus violaceus, Propomacrus cypriacus, Agathidium pulchellum, Lucanus cervus, Phryganophilus ruficollis, Pytho kolwensis, Rhysodes sulcatus, Oxyporus mannerheimii* Butterflies and moths (Lepidoptera) - *Callimorpha (Euplagia) quadripunctaria, Erebia sudetica, Erannis ankeraria, Arytrura musculus, Dioszeghyana schmidtii, Xestia borealis, Xestia brunneopicta, Xylomoia strix, Apatura metis, Coenonympha hero, Fabriciana niobe elisa, Hypodryas maturna, Lopinga achine, Nymphalis vaualbum, Parnassius mnemosyne, Leptidea morsei, Graellsia isabellae* Grasshoppers (Orthoptera) - *Odontopodisma rubripes, Pholidoptera transsylvanica*

molluscs	*Caseolus sphaerula, Geomalacus maculosus, Discus guerinianus, Elona quimperiana, Drobacia banatica, Kovacsia kovacsi, Leiostyla cassida, Leiostyla lamellosa, Vertigo angustior*
plants	Bryophyta - *Bryhnia novae-angliae, Bryoerythrophyllum campylocarpum, Buxbaumia viridis, Cephalozia macounii, Cynodontium suecicum, Dichelyma capillaceum, Dicranum viride, Distichophyllum carinatum, Echinodium spinosum, Herzogiella turfacea, Orthotrichum rogeri, Plagiomnium drummondii, Scapania carinthiaca, Tayloria rudolphiana, Mannia triandra* 77 vascular plant spp

Annex 2 - Understanding the Consequences of Biomass Use for Energy – Land Use Questions - *Notes on key studies and models*

IRENA (IRENA and European Commission, 2018) is the most up to date study looking at future RES demand and costs. It was used in part to justify the extension of the RED target for 2030 from 27% to 32% and hence also models an increased RES target. Other study results are generally based on the Commission proposals ie 27% RES accompanied by the proposed energy efficiency and GHG reduction targets, or model similar levels. The 2015 ILUC study only models to 2020 and does not consider the post 2020 period.

PRIMES modelling – an economic model that differentiates demand for RES that includes a biomass supply element. PRIMES data in terms of demand for biomass based on specific scenarios were used as a basis for the GLOBIOM based modelling for biofuels and biomass (Forsell et al, 2016a; Forsell et al, 2016b). The PRIMES data generates the demand for biomass based on the RES targets and the GLOBIOM analysis is intended to look at how this spatially is distributed and resulting supply side information – although the detail on specific spatial impacts is still more limited.

Green X model - looks at supply and demand generating national cost curves across the EU 28 estimating RES differentiation – used in the BIOSUSTAIN report (PwC et al, 2017), this also used policy baselines and information from PRIMES but in combination with other data generated by Green X to look at supply and demand of RES in MS and within the context of supply and demand for biomass for energy. While info is generated about the potential scale of land impacts this is inferred qualitatively based on data re nature of biomass supply.

To note that there are quite different estimates delivered by the different models re supply of bioenergy see below figure from the Biosustain report (PwC et al, 2017), comparing sources of bioenergy supply between the two models using the same policy scenario.

Table 2-4 Comparison of bioenergy supply in 2020-2030 between PRIMES and Green-X EUCO27 scenario

Comparison of PRIMES and Green-X bioenergy supply*		2020		2030	
		PRIMES	Green-X	PRIMES	Green-X
Crops	Mtoe	20.8	27.0	22.7	26.2
Agricultural residues	Mtoe	18.0	7.1	21.2	21.1
Forestry (excl. black liquor)	Mtoe	52.4	96.6	54.4	107.1
Black liquor	Mtoe	17.2	4.8	17.5	6.3
Waste	Mtoe	69.4	37.6	75.3	43.9
Domestic TOTAL	**Mtoe**	**177.9**	**173.1**	**191.0**	**204.6**
Solid biomass imports	Mtoe	18.3	13.1	21.4	15.9
Biofuel imports	Mtoe	2.4	4.8	6.8	5.4
Imports TOTAL	**Mtoe**	**20.7**	**17.9**	**28.1**	**21.3**
Bioenergy TOTAL	**Mtoe**	**198.6**	**191.0**	**219.1**	**225.8**

*Note: * incl. certain waste streams in the case of PRIMES*

To further note that there are wider trends within modelling exercises/within assumptions on land use that will influence how significant they consider the impacts of biomass use for energy to be. For example, analysis using GLOBIOM assumes a certain shift in overall cropland and productive land based on FAO estimates. This implies a decline in cropland up to 2030. See figure below extracted from the ILUC study (Valin et al, 2015)

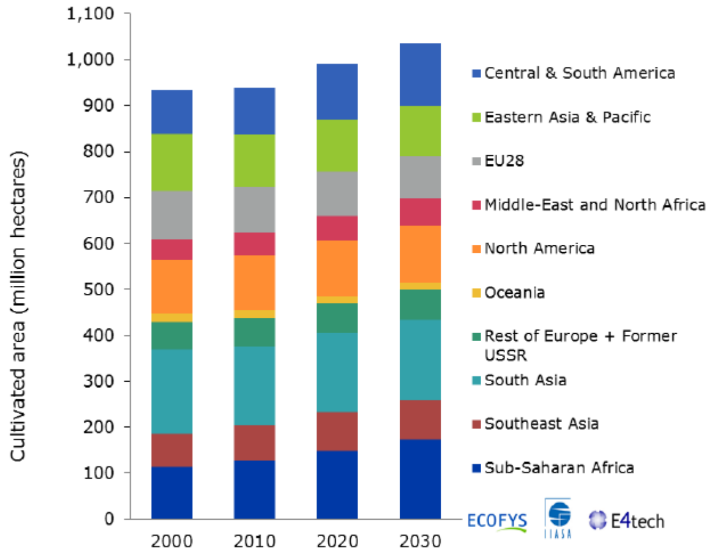


Figure 8: Total cultivated area per region in the baseline projections. Source GLOBIOM

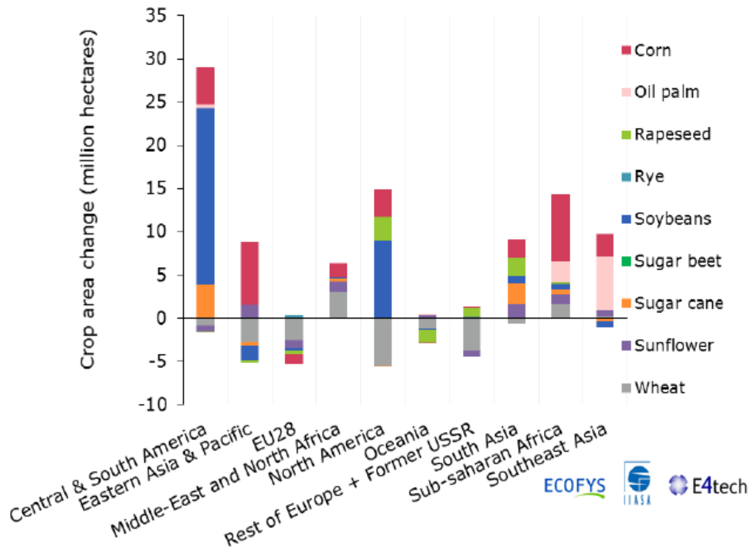


Figure 9: Crop area change in the baseline between 2010 and 2030. Source: GLOBIOM

Extracts of Data from Key Studies

Data/Study		Assumptions	Land Use issues/biomass demand conc (judgement unless noted)
Demand for biomass for energy - only	**IRENA study** (IRENA and European Commission, 2018)looked at biomass use to 27% RES but importantly also to delivery beyond 27% by 2030. Identified that BAU would delivery 24% RES, but that you could deliver cost effectively 34%. This 34% implied a decline in the percentage of RES coming from biomass but importantly a real world expansion in demand for biomass for energy focused on transport biofuels (note that the EU rules now potentially restrict what can be used both in terms of crops, ILUC impacting fuels and promotion of advanced fuels, Remap doesn't take this into account as already completed before rules imposed); within district heating; within heating and cooling for industry. Note that this is additional increases above the reference scenario that in their modelling delivered approx. 24% RES and already implied an increase in biomass use across all sectors from the 2010 baseline. *Under the Reference Case, biomass demand is expected to grow substantially to reach 9.6 EJ by 2030. If the potential of all REmap Options is implemented, biomass demand would grow further to reach 12.2 EJ, roughly a twofold increase from 2010 to 2030' ' A study prepared for the European Commission* (PwC et al, 2017) *estimates that the supply potential of domestic bioenergy in the EU-28 by 2030 ranges from 14.1 EJ to 16.4 EJ.'*	costs and market penetration of biomass technologies versus cost and market penetration of alternative RES technologies, scale of efficiency in the energy supply chain (as targets for RES are based on the proportion of energy from RES sources)	Scale of demand for biomass energy differentiated by sector potentially but likely based on energy units rather than physical units of biomass. Implies a continued expansion in biofuels. Although note this was drafted pre new RED II rules, hence expansion of biofuels would likely be focused on lignocellulosic materials either from forest residues, wood materials, agricultural residues and lower ILUC/Higher GHG based conventional fuels ie based on sugar beet, sugar cane and possibly some of the lower GHG biodiesel fuels ie sunflowers. Focus on industry and buildings is generally in relation to heat hence primarily likely wood chip or pellets. Again, new RED II rules will be important in determining sourcing.
Demand for biomass with supply side and land demand impacts – consequences assumed on a qualitative basis	**BIOSUSTAIN** report (PwC et al, 2017) - The Green-X Model allocates biomass feedstock to feasible technologies and sectors in a fully internalised calculation procedure. For each feedstock category, technology choices and their corresponding demands are ranked based on the feasible revenue streams as might be available to a potential investor under the conditioned, scenario-specific energy policy framework that may change annually. Despite the continuous increased use of bioenergy up to 2030, specifically in heating & cooling and in the electricity sector, their percentage share in final energy production coming from renewables is declining modestly (cf. Figure 2-1 (bottom)) from around 61% by 2014 down to about 50% by 2030. As per other studies proportion of bioenergy is reducing (although not a decline in total volumes). The Scenario for 27% RES and 27% energy efficiency resulted in 146.5 Mtoe of biomass within the RES energy mix (22.7 Mtoe in electricity –	Based on the proposed 27% targets for 2030 and linked goals for GHG reduction and energy efficiency of 27% with additional sensitivity analysis looking at the impacts of restricting eg foresty inputs through different policy mechanisms aimed at applying restrictions to deliver sustainable production.	The Green X model does look at land use – see figure d which sets out area of land anticipated to be needed for domestic energy crops. Note that land used for energy crops is estimated to increase to 8Mha in 2020 compared to 6 Mha in 2012 driven by increased demand for food based energy crops; however, between 2020 and 2030 land required for food and feed based crops declines whilst land cultivated with lignocellulosic energy crops is projected to increase 7 fold from 0.14 Mha in 2013 to 1 Mha in 2030. Important to note that the amount of land areas required for cultivation of energy crops in restricted by the model to so-called 'surplus land' ie not need for other purposed including food production. Land available for lignocellulosic crops is estimated

	5.3 biogas, 14.6 solid biomass, 2.8 biowaste) 94.6 Mtoe in heat (biogas 2.7, 65.8 direct bioenergy use, other figures were derived heat); 17.4 Mtoe for bioenergy in transport including 10.2 conventional; 5.2 advanced fuels). The analysis also suggested that the majority would be sourced from domestic bioenergy feedstocks the vast majority from forestry sources – See figure a and b below for breakdown of sources based on the EU delivering 27% RES. Note that scenarios that restricted forest biomass inputs eg imposing caps on Roundwood tended to drive a increased use of agricultural land for biomass which pushed up costs so while it drove expansion into agricultural land of say SRC this was higher cost that use of woody material hence reduced overall use of biomass as well. The study also looked at intra EU trade indicating important suppliers on the EU market place – Agropellets made from agricultural residues (straw, sunflower husk etc) make up 10% of EU pellet production (617 ktoe) with production in Poland (198ktoe), Romania (52 ktoe).		based on estimates in Elbersen et al 2015. In additional bioenergy feedstocks were excluded from being produced on high biodiversity areas (in line with the RED definition) and high carbon stock land. Total land available for energy crop cultivation on surplus land in the EU is estimated to increase to 22.5 Mha by 2020 and 24 Mha by 2030. It is noted in the report itself that this distinction is artificial and doesn't reflect market forces or regulations. They conclude no land use impacts given the difference between surplus land and land need for energy crops; the decline in the use of food and feed based crops and the increase in use of lignocellulosic crops – which they note as having potential biodiversity benefits. Although they note that impacts are location specific. The rest of the discussion focused on international impacts linked to ILUC. The report does look at policy options for example that impose sustainability rules for heat and power (ie what has now been done for the RED), very limited discussion on land use impacts, the main being noted in relation to the extension of rules protecting agricultural land to biogas production. The report also analysed imposing forest restrictions ie risk based approaches and SLM criteria for forestry. For these it is considered that these would potentially reduce forest production and hence lead to increased production of energy crops on agricultural land, specifically woody crops.
Fixed demand and scenario's looking at variable land use consequences	***RECEBIO I*** (Forsell et al, 2016a) – GLOBIOM based analysis of heat and electricity demand impacts, no analysis of biofuels consequences - – Looks at biomass for heat and electricity based on triggering an assumed level of demand for biomass based on estimates emerging from the POLES/PRIMES modelling and the EU Reference Scenario for 2014 . The baseline scenario assumes that the targets on RES and GHG reduction to 2020 exist but that no new policies are adopted to 2030 with RES delivering approx. 24% of gross final energy consumption. Under the scenario biomass for energy and for materials	Uses the baseline of no specific additional RES/Climate policies post 2020. Looks then at an enhanced GHG emission reduction scenario (REDU) that assumes 40% GHG emission reduction and	Under both the emission reduction and baseline scenario a significant increase in the level of SRC in terms of volume and land area was noted - from 0.4 in 2010 to 66 million m3 by 2050 and from 10 000 ha in 2010 to 3.4 million ha in 2050 under the Baseline, and to 161 million m3 and 8.9 million ha in 2050 in the EU Emission Reduction scenario. Although it is noted that SRC development has in the past been problematic and sensitivity analysis showed that in the absence of this expansion increased

	increases significantly in particular to 2020 and then increases at a slower pace to 2050. Should be noted that this was completed based on PRIMEs numbers for the impact assessment 2014 re bioenergy deployment determining the demand for biomass which was then pushed through the model. These were revised hence RECEBIO II work. It should be noted that the baseline scenario shows a clear increase in wood use both for energy and material use to 2050. The increased demand for wood biomass is seen to lead to an intensification in the use of forests in the EU28. There is an expansion in the area of used forest in Europe. There is also a significant expansion in the use of SRC both in terms of volume consumed and area of land devoted to production. These expansions in used forestry and land devoted to SRC lead to a decline in the area of unused forest (most notably leading up to 2050) and a more significant decline in the area of 'other natural land'. Under the emission reduction scenario, that assumes higher demand levels of GHG savings up to 2030 there is actually a lower demand for biomass, compared to the baseline, associated with energy efficiency, however, beyond 2030 harvest levels increase beyond the baseline. Up until 2030 the main driver for forest harvest is considered to be material use. Beyond 2030 high bioenergy demand under the Emission Reduction Scenario has a clear impact on the overall forest harvest level. After 2030, the increasing harvests of wood for direct energy production is expected to become the main driving force for the increasing forest harvests in the EU.	26.4% RES deployment and increased energy efficiency plus a target of 80% GHG emission reductions by 2050. Also completes analysis looking at scenarios where you see increase in demand in the rest of the world in addition to EU.	quantities of Roundwood and imported pellets would likely be used. The analysis also included a scenario where by the rest of the world also increases their demand for biomass inhibiting the ability for the EU to import biomass feedstocks for energy and as a consequence focusing effort more on EU production. The results show that, with an increased RoW bioenergy demand, net EU import of wood pellets is only 39 million m3 in 2050, 25% less than in the EU Emission Reduction scenario. In addition, also EU roundwood imports decrease by more than 20%. This puts more pressure to the development of the SRC sector in the EU: in this scenario, the production of SRC in the EU28 is the highest of all scenarios at 172 million m3 in 2050 (a 7% increase to the EU Emission Reduction scenario). Material production levels stay at almost the same level as in the EU Emission Reduction scenario. However, as EU roundwood imports decrease, the domestic forest harvest level increases to 718 million m3 in 2050 (14 Mm3 higher than in the EU Emission reduction scenario, and 162 Mm3, or 29%, higher than in 2010). The key changes in land use already under the Baseline scenario are an increase in the area of cropland (The cropland area is increasing by 14 Mio ha from about 106 Mio ha in 2010 to 120 Mio ha in 2050) and used forest (used forest increases by 19 Mio ha from 105 Mio ha in 2010 to 124 Mio ha in 2050 as a result of expansion of forest area through afforestation and a reduction of the deforestation rate; total forest area increases by total forest area is increasing by 14 Mio ha from 154 Mio ha to 168 Mio ha; Unused forests remain rather stable over this time period (declining slightly from 48 Mha in 2010 to around 42 Mio ha in 2050) in the EU, driven to some degree by the increased demand for SRC and increased forest harvest level, respectively. Following this development, we see a decline in the area of unused forest and, most significantly, other natural land. These trends are seen to be enhanced further under the Emission Reduction scenario with, by 2050, higher amount of land being used in the EU for SRC,

			and lower amounts of other cropland and other natural land (including abandoned cropland and grazing land) and grazing land (2030 and 2050). When comparing the Baseline and Emission Reduction Scenario, the total forest area (sum of used and unused forest) does not differ significantly; however, there are comparably large shifts within the forest, converting unused forest to used forest.
Fixed demand and scenario's looking at variable land use consequences	***RECEBIO – follow up study*** – (Forsell et al, 2016b) Based on the same GLOBIOM modelling effort except the analysis is based on updated scenarios (REDU2) based on PRIMES scenario EUCO 27 – this assumes that the EU achieves at least 40% GHG emissions (compared to 1990) reduction target (with the split ETS/non-ETS reducing by 43%/30% in 2030 compared to 2005), a 27% share of renewables and a 27% energy efficiency target. The primary difference between the REDU2 and the REDU is that updated projections for bioenergy demand taken from PRIMES 2016 EUCO 27. Most importantly, in REDU2 the demand for SRC and lignocellulosic crops is fixed at the same levels as in PRIMES 2016 EUCO 27 development (instead of being price-elastic as in REDU), and the share of roundwood in wood pellets is assumed to be 75% (instead of 0% in REDU). The largest difference between the REDU2 and REDU is the amount of short rotation coppice (SRC) domestically produced within the EU: in REDU, the SRC production was modelled endogenously and was in competition with other feedstocks. In REDU2, SRC was fixed to the PRIMES 2016 EUCO 27 scenario development. As a result, SRC production in REDU2 increases much more slowly until 2030 than in REDU. However, thereafter SRC increases very rapidly in REDU2, reaching 280 Mm3 by 2050 – more than 70% increase compared to REDU in 2050. Because the development of SRC **between 2010 and 2030** is projected to be slower in REDU2 than in REDU, the amount of forest biomass demanded for energy use is conversely 22% higher in 2030 in REDU2 than in REDU. This higher demand for non-SRC woody biomass for energy affects the EU import of wood pellets from RoW, which doubles in REDU2 from the level of REDU, and roundwood combusted directly for energy, which is almost doubled in REDU2 as compared to REDU in 2030.	Based on assumptions both in terms of RES and overall biomass use for heat and power, but also specific values in terms of demand for SRC and lignocellulosic crops generated by PRIMES modelling under the 2016 EUCO 27 scenario	The REDU2 results show a clear increase of wood used for both material and energy production between 2010 and 2050. On the bioenergy side, the results show a considerable increase over time in the use of imported pellets (from 10 Mm3 in 2010 to 70 Mm3 in 2050) and EU domestic production of SRC (from negligible amounts in 2010 to 280 Mm3 in 2050, following the PRIMES estimates). Additionally, the rapid increase of bioenergy demand is seen to also lead to large quantities of domestic roundwood combusted directly for bioenergy production (in the form of logs, chips, or pellets) (25 Mm3 in 2050). In other words, the bioenergy demand increases to an extent where stemwood that is of industrial roundwood quality (mainly pulpwood quality). The increased use of biomass for energy and material is expected to lead to intensification in the use of EU forests. The forest harvest level in the REDU2 scenario is seen to reach a level of 660 Mm3 by 2050 (12% higher than in 2010). Land use in the scenario is also characterized by the increase of SRC: the land area used for SRC expands from almost zero to 15 Mha in 2050. We detect also an increase of the total forest area within the EU by almost 14 Mha in 2050 compared to 2010. Both of these land use changes are found to mostly result from a change from other natural land (abandoned cropland, unused grassland, etc.). Under the REDU2 while cropland (excluding SRC) and grazing land remain relatively stable at 107-112 Mha and 53-56 Mha without a clear trend over the course of the 40 years of simulation, there is a large increase of SRC area from zero to almost 15 Mha in 2050.

			Similarly, unused forest area remains stable at around 40-45 Mha until 2050 but used forest area increases from 105 in 2010 to 130 Mha in 2050. Both forest and SRC area expansion comes at the expense of other natural land. Between 2010 and 2050, 35 Mha of this land category is converted to forest and SRC, cutting this land category by more than 50%. Note that this is in contrast to the REDU results that showed conversion of un-used to used forests. It should be noted, however, that when other restrictions are applied to the sourcing of biomass material ie under the LAND scenario (that prevents sourcing of biomass from high carbon stock and highly biodiverse land) and in the RWCAP scenario (that caps use of roundwood for energy) see a significant decline in unused forest and an increase in used forest within the EU by 2050.
Fixed demand – LUC impacts associated with individual biofuel crops	GLOBIOM – Land Use Change impact of biofuels consumed in the EU (Valin et al, 2015)- Biofuels only but covers both conventional and advanced feedstocks– Intended as an input to the ILUC debate, this analysis looks at the land use change consequences by different demand for biofuels in 2020 based on national estimates of demand in National Renewable Energy Action Plans and in scenarios where the level of in particular conventional fuels is capped at 7%. The study looks at land use change in total rather than just indirect land use change, as it is often unclear whether the change is generated directly or indirectly and due to the link between direct and indirect changes (ie the former generating the latter elsewhere). The study focused primarily on the emissions associated with land use change, however in doing so had to consider scale of land use change associated with the different commodities and was the first study to do so for advanced crops. It was not focused on biodiversity but does provide detail on land use impacts that could be used to support understanding of the impacts of biofuel feedstock use.	Assumes that biofuels demand increases by a fixed proportion above the 2008 baseline by 2020 based on the National Renewable Energy Action Plans and exacts a land use impact by triggering a shock of additional biofuel demand and examining the land use and specifically the GHG impacts associated. Not focused on biodiversity.	Unlike many studies it specifically looks at the land use consequences, albeit to the end of identify LUC associated emissions. However, within the analysis it includes both LUC impacts associated with an overall change in biofuel demand at 2020 of 6.2% increase in consumption compared to 2008 levels and a specific increase in demand for individual biofuel feedstocks with a shock of an increase in 1% of demand for biofuels to be delivered via that feedstock (123PJ). This study therefore supplies some information of on which land use types conversion falls depending on the crops selected – see figure f. It also provides more detail on whether this conversion is likely to be within the EU 28 or external (ie in third countries) – see figure g. It is important to note that from figure g increase in crop based fuels comes largely in Europe from abandoned land or other natural land ie wheat, maize, maize silage, barley, sunflower oil, rape seed oil and for cereal straw (although total LUC in the latter is much lower overall). Perennial (ie miscanthus and switch grass) crops were also being seen primarily on natural land and abandoned land with SRC on cropland, natural land and

			abandoned land. This natural or abandoned land will sit outside natura sites (as Globiom generally excludes use of materials from protected areas). However, from a biodiversity perspective is likely to be important habitats or refuge habitats in agricultural areas. In terms of specific land areas under the 2020 scenario that imposes a 7% cap on conventional biofuels land expansion leads to 6.7 Mha of land conversion globally, of which 5.2 Mha are used for additional cropland and 1.5 Mha for short rotation coppice. In the EU, cropland expands by only 1.8 Mha, half at the expense of abandoned land and half through other natural vegetation.

Figure a – Modelled bioenergy supply in 2020 to 2030 in Green-X scenario. Extract from Biosustain study (PwC et al, 2017).

Figure 2-2 Bioenergy supply in the period 2020-2030 in the Green-X EUCO27 scenario

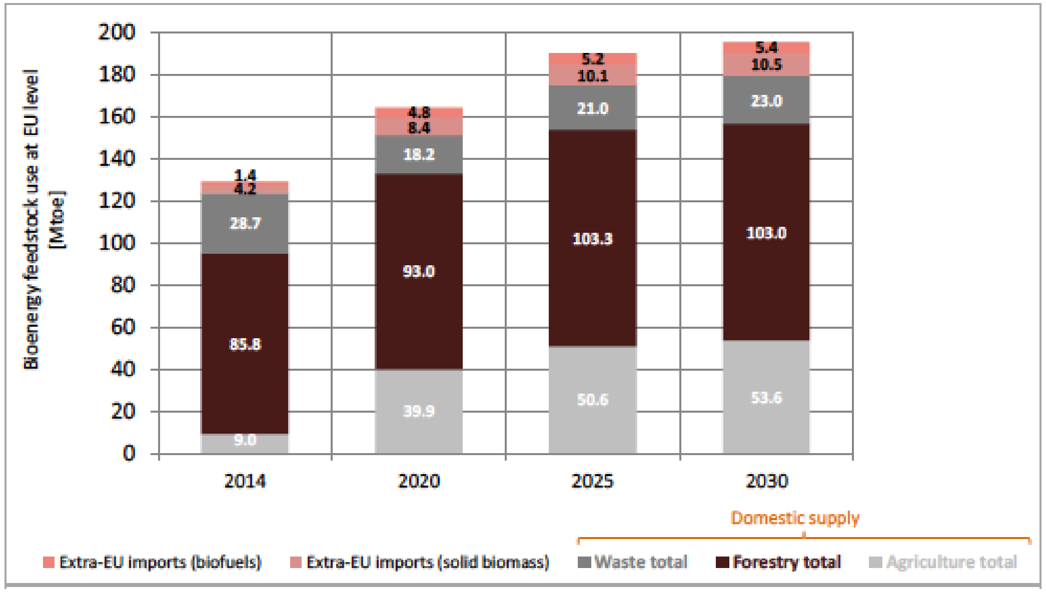


Figure b – developments in land used related to energy crop cultivation in the EU 28. Note that the PRIME model estimate differs, and this is mainly put down to a much higher starting estimate of land already under production for bioenergy of 11 Mha in 2015. Extract from Biosustain study (PwC et al, 2017).

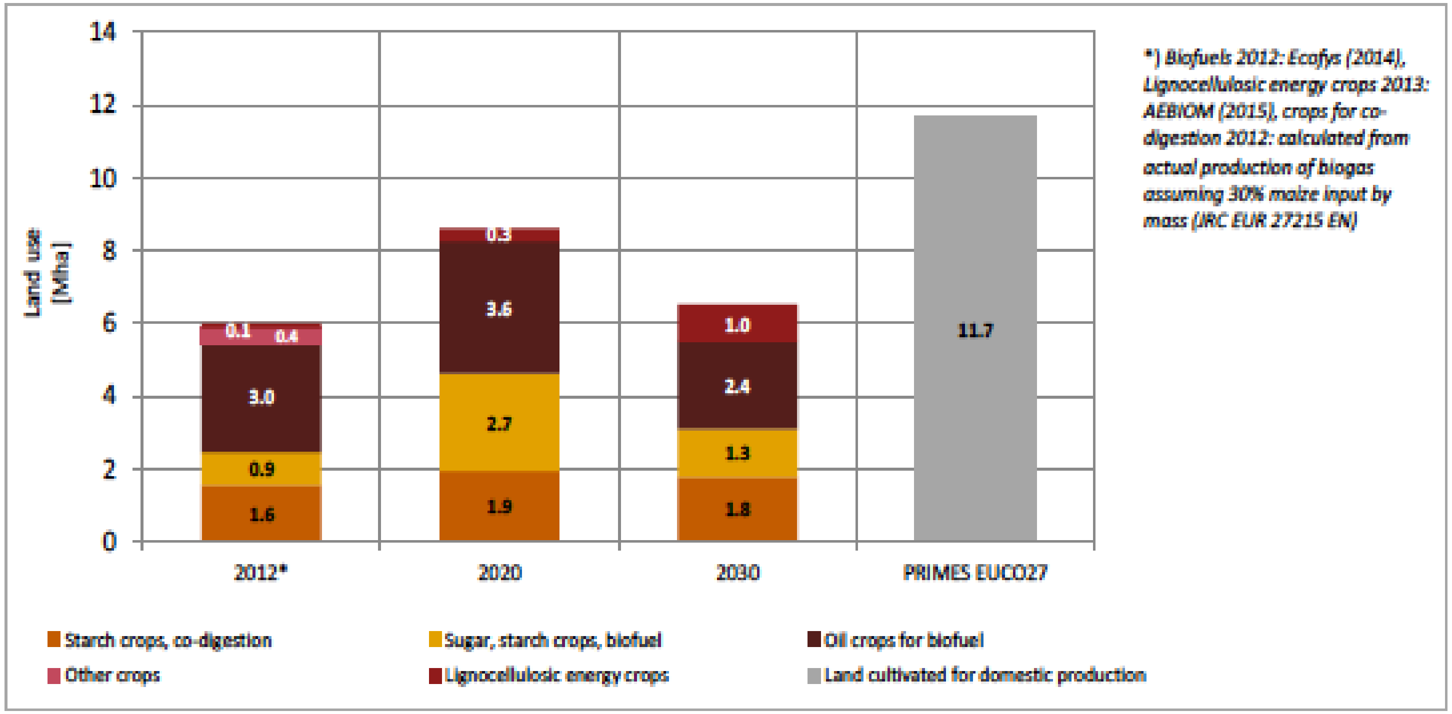


Figure c – Change in land use patterns for energy crops when forestry risk-based criteria are imposed. This may imply such a change as the new RED includes these – however, whether this is a true reflection of change depends on how accurate the modelling of risk-based criteria actually can be given they didn't know the criteria to be applied and the limitations of imposing such restrictions through the models (which normally simply restricts some sources of biomass material again based on assumptions re policy impact). Extract from Biosustain study (PwC et al, 2017).

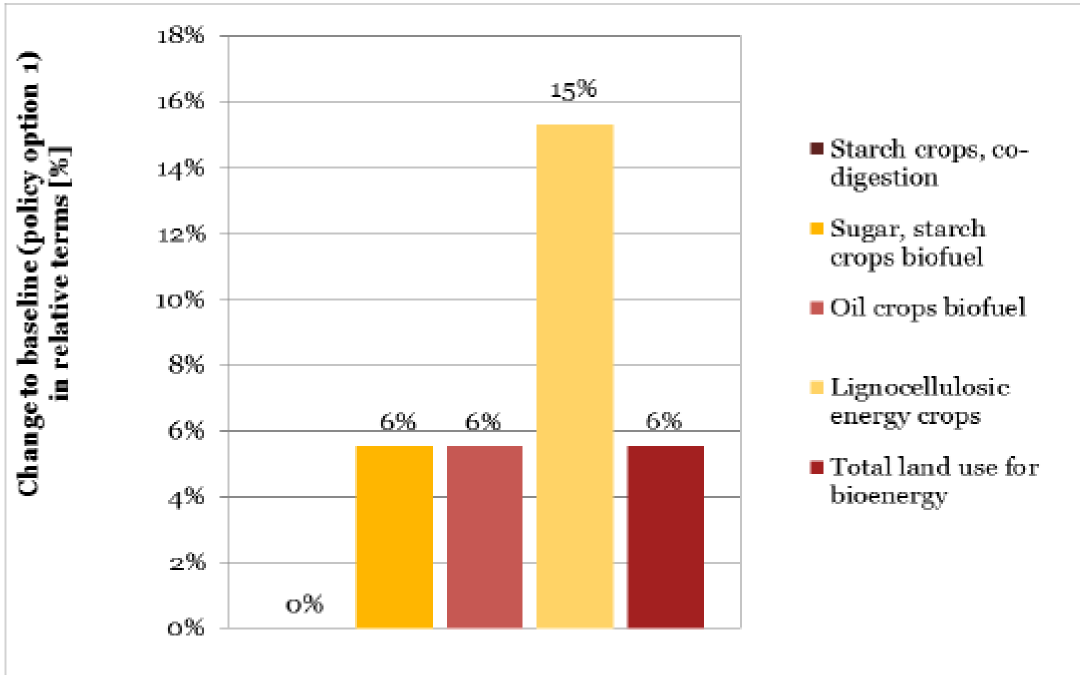
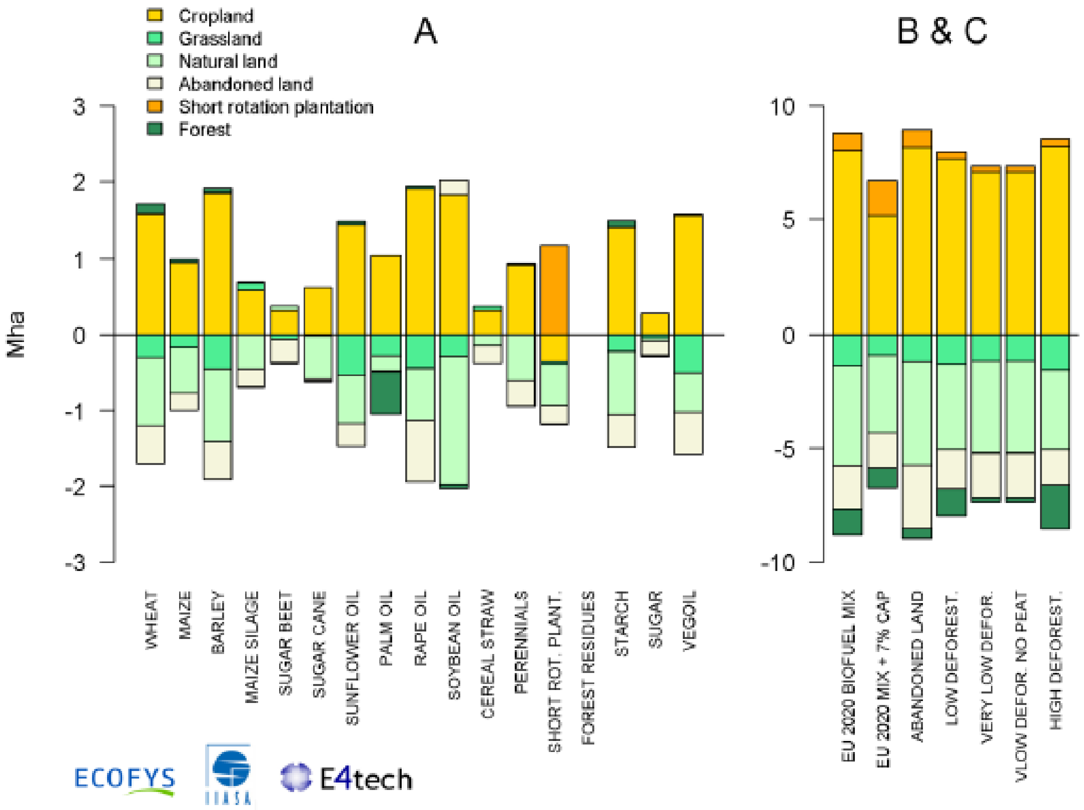


Figure d – Specific land use consequences associated with 'shocks' to the system to increase in A) consumption of biofuels produced from each feedstock by 1% of the total biofuel consumption in the baseline (ie 2008 levels) (123PJ) and in B) scenarios for 8.6 % biofuel use in 2020 and EU 2020 mix 7% - imposing a cap on food and feed-based fuels. Extract from the ILUC study (Valin et al, 2015).



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