Spatially explicit LCA analysis of biodiversity losses due to different bioenergy policies in the European Union

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Abstract

In this study, the potential global loss of species directly associated with land use in the EU and due to trade with other regions is computed over time, in order to reveal differences in impacts between the considered alternatives of plausible bioenergy policies development in the EU. The spatially explicit study combines a life cycle analysis (LCA) for biodiversity impact assessment with a global high resolution economic land use model. Both impacts of domestic land use and impacts through imports were included for estimating the biodiversity footprint of the member states of the (EU28). The analyzed scenarios assumed similar biomass demand until 2020 but differed thereafter, from keeping the growth of demand for bioenergy constant (CONST), to a strong increase of bioenergy in line with the EU target of decreasing greenhouse gas (GHG) emissions by 80% by 2050 (EMIRED) and with the baseline (BASE) scenario falling between the other two.

As a general trend, the increasing demand for biomass was found to have substantial impact on biodiversity in all scenarios, while the differences between the scenarios were found to be modest. The share caused by imports was 15% of the overall biodiversity impacts detected in this study in the year 2000, and progressively increased to 24% to-26% in 2050, depending on the scenario. The most prominent future change in domestic land use in all scenarios was the expansion of perennial cultivations for energy. In the EMIRED scenario, there is a larger expansion of perennial cultivations and a smaller expansion of cropland in the EU than in the other two scenarios. As the biodiversity damage is smaller for land used for perennial cultivations than for cropland, this development decreases the internal biodiversity damage per unit of land. At the same time, however, the EMIRED scenario also features the largest

outsourcing of damage, due to increased import of cropland products from outside the EU for satisfying the EU food demand. These two opposite effects even out each other, resulting in the total biodiversity damage for the EMIRED scenario being only slightly higher than the other two scenarios.

The results of this study indicate that increasing cultivation of perennials for bioenergy and the consequent decrease in the availability of cropland for food production in the EU may lead to outsourcing of agricultural products supply to other regions. This development is associated with a leakage of biodiversity damages to species-rich and vulnerable regions outside the EU.

In the case of a future increase in bioenergy demand, the combination of biomass supply from sustainable forest management in the EU, combined with imported wood pellets and cultivation of perennial energy crops, appears to be less detrimental to biodiversity than expansion of energy crops in the EU.

Keywords: biodiversity damage, bioenergy, land use, perennial energy crops, forestry, EU footprint, trade.

1. Introduction

The EU recently updated its targets for bioenergy use in order to reach a 40% reduction of greenhouse gas emissions by 2030 compared to 1990 levels (European Commission, 2016; European Parliament and Council of the European Union, 2016). Within the refined target, 27% of the total energy consumption is expected to be provided through renewable resources by 2030 (European Parliament and Council of the European Union, 2016). Bioenergy currently provides 59% of the renewable energy consumed in the EU (Eurostat 2016). In addition to the increased renewable energy consumption targets, awareness of the sustainability of bioenergy supply is also on the rise.

An increase in the demand for woody biomass in Europe is expected to lead to an increased harvest level in currently managed forests through elevated tree part utilization, expanding forest area, and short rotation coppice plantations, as well as increasing wood imports from other regions, and/or increasing wood supply from outside the forests (Mantau et al. 2010, Lauri et al. 2014, Forsell et al. 2016, Schelhaas et al. 2006).

Depending on the different point of demand, the biomass can assume different shapes, for example, solid wood fuels such as wood pellets or it can be converted into biofuels.

In this context, many European environmental non-governmental organizations (NGOs) argue that without appropriate sustainability criteria for most biofuel production, the increased use of woody biomass may lead to negative environmental impacts (Obersteiner et al. 2018).

Therefore, increased use of woody biomass to replace fossil fuels is likely not a side-effect free solution to climate change problems. Increased biofuel production could lead to increased loss in biodiversity and may also indirectly impact food security through possible increases in food prices or further competition for land use (Söderberg & Eckerberg 2013). Liquid biofuels can be divided into two categories: first-generation biofuels made from the sugars and vegetable oils of arable crops, and second-generation biofuels made from ligno-cellulosic biomass, such as woody biomass. The EU has reported that the negative impacts of first-generation biofuels, such as deforestation, competition with food production, and indirect land use change, provide motivation for a preference for second-generation biofuels from ligno-cellulosic biomass (European Parliament and Council of the European Union, 2015). Accordingly, both the EU and the USA have actively been promoting a revision of their policies with a shift away from first-generation biofuel crops such as corn, sugarcane, and oilseeds towards cellulosic biofuels that utilize the woody or fibrous parts of plants (Baumber 2017).

One of the global criteria for sustainable development (i.e., Sustainable Development Goal 15) is to ‘Protect, restore, and promote the sustainable use of terrestrial ecosystems, sustainably manage forests, combat desertification, halt and reverse land degradation, and halt biodiversity loss’ (UN 2015). The EU has also recognized the importance of biodiversity explicitly, and adopted a strategy to halt biodiversity loss by 2020 (European Commission 2011). This strategy includes, among others, targets to improve the conservation status of habitats and species, and to improve and restore ecosystems and ecosystem services wherever possible. Land use and its changes are considered the main drivers for biodiversity loss in terrestrial ecosystems (Pereira et al. 2010), and the general consensus is that more land protection is required to preserve global biodiversity (Heller and Zavaleta 2009). With this connection, it is evident that the impacts of future bioenergy policies need to be assessed in light of their impacts on the land use, land-use change, and forestry (LULUCF) sector, as well as their related impacts on biodiversity.

Managing policy trade-offs connected to the LULUCF sector and biodiversity is complicated by the interconnected nature of global energy, food, feed, and fiber markets. While some impacts of increased bioenergy production are direct (observed in the areas where biomass is produced), others are indirect, affecting land use change and the supply of food, feed and fiber in other distant locations (Berndes et al., 2011). Through indirect land use change, the impacts of EU policies are also transferred to highly vulnerable habitats in other regions such as Asia or South America (e.g., Rivas Casado et al. 2014, Britz & Hertel 2011). That is, the interaction between bioenergy supply and larger global systems leads to indirect consequences on the globe beyond the direct effects connected to the bioenergy production chains (Elbersen et al. 2013). The rapidly increasing demand for biofuels, driven in part by EU policies, is a clear example of this due to the global nature of biofuel markets. In this case, the reported effects were damages to biodiversity and ecosystem services provision through both direct and indirect land use changes (Holland et al., 2015).

Previous studies have measured the impacts of the LULUCF sector on biodiversity in the EU under different bioenergy policy scenarios (Eggers et al. 2009, Rivas Casado et al. 2014, Schulze et al. 2016). These approaches assess the suitability of different land uses as habitat for different species. However, as these studies only measure change in biodiversity related to change in land use within the area directly impacted by a policy, they fail to account for changes in biodiversity related to two other vital processes namely: i) changes in biodiversity related to the intensity of land use and forest activities, and ii) changes in biodiversity in areas that are only indirectly affected by the policy, for example the impacts on biodiversity outside the EU as a result of market effects and international trade of food, feed and fiber commodities.

Recently, global databases containing responses of species to different land uses and intensities of management have been made available (Hudson et al 2014; Schipper et al. 2016). These databases have allowed for a regionalized quantification of biodiversity losses consistent with a global framework (Newbold et al. 2015, Chaudhary et al. 2015).

The development of spatially explicit biodiversity indicators within LCA has progressed substantially in the last years (for a review see Curran et al., 2016). Among the most notable developments are the methods of de Baan et al. (2012, 2013), which were the first to quantify local, regional, and permanent biodiversity loss on a global scale. Following the suggestion of Verones et al. (2013), Chaudhary et al. (2015) developed these approaches further by including more data and weighing regional species loss with a factor combining the rarity and threat level of species. Their work provides impact factors that measure biodiversity loss in units of global species extinctions at a steady state, that is, the number or fraction of species that are committed to extinction in the long term as a consequence of land use for six land use classes and 804 ecoregions. The joint Life Cycle Initiative (2016) between the United Nations Environment Programme (UNEP) and the Society of Environmental Toxicology and Chemistry (SETAC) tentatively recommended the method of Chaudhary et al. (2015) as best practice for the assessment of land-related impacts on biodiversity loss. This method has been used to assess the biodiversity impacts of global agriculture and forestry (Chaudhary et al. 2016a) and also for global trade (Chaudhary & Kastner 2016). However, none of the existing studies have assessed biodiversity loss of prospective land use scenarios under different policies.

In this paper, we set out a global framework that is able to jointly assess and analyze the biodiversity implications of policies related to: direct land use change, changes in intensity in land use and forestry, and in-direct land use effects.

We build on the recent development of biodiversity indicators within LCA, and provide a spatially explicit analysis of LULUCF driven biodiversity loss from different European policies in the bioenergy sector. In our analysis, the biodiversity loss factors are coupled with the results of the Global Biosphere Management Model (GLOBIOM) – a high resolution economic model providing prospective land use scenarios that also allow us to analyze impacts on other regions and on international trade (Havlık et al. 2011, Havlik et al, 2014). Three alternative bioenergy policy scenarios are considered in the EU during the period from 2000 to 2050. The potential global loss of species directly associated with land use in the EU and due to trade with other regions is computed over time, in order to reveal differences in impacts between the considered alternatives of plausible policy development in the EU.

2. Material and methods

2.1 Approach for assessing land-use related impacts of biodiversity loss

For the assessment of biodiversity loss from LULUCF, the life cycle impact assessment method "potential loss of global species (PSLglo)" was used. The approach quantifies the percentage of global species lost at a steady state, thereby providing an indicator of global extinctions that will result as a consequence of LULUCF. Species loss is quantified using the countryside species area-relationships (SARs). In contrast to the original SARs, it takes into consideration that some species will also survive in anthropogenically transformed land, depending on their affinity. Regional species loss is further weighted with the total range and threat level of species to provide an indicator of global species extinctions (i.e., global species equivalents lost per m^2). The PSLglo method provides characterization factors (CFs i.e., the factors indicating the biodiversity damage caused by the unit area of a particular land use in a particular region) for six land use types (annual crops, permanent crops, pasture, urban areas, extensive forestry, intensive forestry), four vertebrate taxa (mammals, birds, amphibians, and reptiles), vascular plants, and 804 ecoregions. Ecoregions are chosen as spatial units containing distinct communities of species, and their boundaries approximate the original extent of natural ecosystems prior to major land use change (Olson et al. 2001). Taxa aggregated CFs for each land use type per ecoregion are also available in the unit of potentially disappeared fraction of global species (PDF/m^2) (UNEP-SETAC 2016, Chaudhary et al. 2015). To get from the unit "global species equivalents lost" to PDFs, the former is divided by the total number of existing species on earth, for each taxonomic group, thus denoting the fraction of global species that is projected to go extinct. PDFs of various taxonomic groups are then aggregated by calculating a weighted average, following the procedure documented in UNEP-SETAC (2016). In this paper, we used the "marginal" characterization factors reported in UNEP-SETAC (2016).

2.2 Biodiversity impact of future land-use scenarios

The regionally specific CFs were combined with land use maps of annual crops, permanent crops (i.e., miscanthus and short rotation energy plantations), pasture, and managed forests in the EU (EU28) computed from the GLOBIOM model under different bioenergy policy scenarios.

The GLOBIOM model is an economic partial equilibrium model of the global forest, agriculture, and biomass sectors with a bottom-up representation of agricultural and forestry management practices (Havlik et al. 2011, Havlik et al, 2014). In this study, the GLOBIOM model was run recursively for 10-year time steps (i.e., the years 2000, 2010, 2020, 2030, 2040, and 2050) for three different bioenergy policy scenarios. The results from the GLOBIOM model were analyzed at the resolution of 246 European administrative units (NUTS2) (supplementary information (SI) 1) and they are presented as land-use maps for the assessment of biodiversity impacts.

The GLOBIOM model covers the following six main land use categories: unused forests, managed forests, cropland (both annual and permanent), pastures, other natural vegetation, and urban areas. However, for the assessment of the biodiversity implications, only changes in managed forests, cropland, and grassland are used. Unmanaged forests and other natural vegetation were considered as the reference ecosystems in each ecoregion, assuming impacts from human modifications were negligible, while urban areas were considered to be out of the scope of analyses in our scenarios.

Managed forests are forests used over a certain period to meet wood demand. These forests are managed for woody biomass production, which implies a certain rotation time, thinning events, and final harvest. The unmanaged forests do not currently contribute to wood supply, based on economic decision rules in the model. However, they may still be a source for collection and production of non-wood goods (e.g., food, wild game, or ornamental plants).

The land allocated to “managed forests” in GLOBIOM was divided between “intensive” and “extensive” management. Area shares of intensively and extensively managed forest in each NUTS2 unit in the EU were calculated according to a European forest management suitability map from Hengeveld et al. (2012). For this purpose, the “combined objective” forests in Hengeveld et al. (2012) were considered to be “extensive forests”, while the “even aged forests”, and “short rotation forests” were classified as “intensive forestry”. The forest land used outside the EU was divided between “intensive” and “extensive” forest according to the shares of roundwood from plantations reported in Jürgensen et al. (2014) for five regions (i.e., South America, Oceania, Asia, Africa, and North and Central America) in the period 2000 to 2010. The projection of expansion rates for plantations from 2010 to 2050 was based on the trends predicted in ABARE & Pöyry (1999) and Jürgensen et al. (2014).

The matching between the land-use spatial units of NUTS2 with the ecoregion-specific characterization factors was done according to equation 1:

$CF_{i,j} = \sum_{g=1}^{n} CF_{g,i} \times p_{g,j}$ (Eq.1)

Where $CF_{g,i}$ is the characterization factor for the land use types i (cropland, permanent crops, extensive forestry, intensive forestry, and pasture), j is an index for NUTS2 units, g is an index for ecoregion, and $p_{g,j}$ is the share of area occupied in the NUTS2 region j by each ecoregion g.

Biodiversity damage $BD_{i,j}$ (species eq. lost) impact due to the different land uses in each NUTS2 was calculated by multiplying $CF_{i,j}$ by the area ($A_{i,j}$) occupied by the different land use types in each of the NUTS2 (in m^2), thus assuming a steady state change in biodiversity as:

$$BD_{i,j} = CF_{i,j} \times A_{i,j} \quad \text{(Eq. 2)}$$

The sum of the $BD_{i,j}$'s from different land uses i provided the NUTS2 level biodiversity damage:

$$BD_j = \sum_i BD_{i,j} \quad \text{(Eq. 3)}$$

We assessed the impacts using the taxa aggregated $CF_{i,j}$'s for each land use type in the NUTS2. This provides the biodiversity impacts in the units potentially disappeared fraction per m^2 of land use (PDF/m^2). The impacts due to land use from the NUTS2 were then also provided on the country level.

$$BD_c = \sum_j BD_j \;\; ; for\ all\ j\ located\ within\ country\ c \quad \text{(Eq. 4)}$$

The PDFs due to forest land use in the EU were divided by the roundwood production from each NUTS2 unit, and a map of impacts as global PDF/m^3 roundwood was obtained. The same indicator (PDF/m^3) was also calculated at the country and EU levels.

Similarly, for permanent crops including willow and poplar short rotation coppices (SRC) and miscanthus, the PDFs due to land use were divided by their respective production (solid m^3) and the PDF/m^3 perennials were obtained at the NUTS2, country and EU levels. For miscanthus, the conversion to solid m^3 was obtained by calculating the amount of biomass (oven dry tonnes) required for achieving the same energy as 1 m^3 of woody biomass.

2.3 Impacts from trade

In the GLOBIOM model, trade is modeled between 30 global trade regions (i.e., 29 regions and the EU28) (SI 2). The model provides the amount of goods traded by the EU28 countries with the other 29 regions on the globe in each scenario and year. In addition, the model computes the amount of goods traded by each EU country with other European countries.

A trade balance was created for the aggregated EU28, as the difference between the import and export of each product (i.e., net import or, in the case of negative values, net export). Similarly, the trade balance was also created for each country within the EU28. The net import to the EU28 was allocated to the member states proportionally, based on the magnitude of their respective net imports for each of the products.

Agricultural products imported from or exported to the EU28 (i.e., barley, dry beans, cassava, chick pea, corn, cotton, groundnuts, millet, palm oil, potatoes, rapeseed, rice, soybeans, sorghum, sugar cane, sunflower, sweet potatoes, and wheat) were classified as “annual crops” for this specific assessment and were all converted to fresh tonnes of biomass. The amounts were divided by the average yields (tonne/ha, SI 3) in each trading region to obtain the average area of crop used to produce the amounts being traded.

The forest products imports/exports accounted for in this assessment included pulp logs, sawlogs, woodchips, and wood pellets. Their imports/exports were all converted to solid m^3. The traded amounts were divided by the average forest increment (m^3/ha, SI 4) in each trading region to obtain the amount of intensive and extensive forestland used.

For the calculation of impacts $BD_{import,a,EU}$ due to net imports from trade region *a* into the EU28 region, the characterization factors $CF_{g,i}$ for ecoregion *g* for land use types *i* were first multiplied by their area share $p_{g,a}$ in region *a*. The resulting characterization factors for each trade region of origin *a* $CF_{a,i}$ were then multiplied by the area demand $A_{a,i,r}$ of land use type *i* for producing one unit of product *r* and the net amount of product t exported from trade region a to the EU28 region $t_{a,EU,r}$. Afterwards, the total biodiversity damage created outside the EU due to net imports of biomass into the EU $BD_{import,r,EU}$ was calculated by summing the damages through imports from all regions of origin (Eq. 7).

$$CF_{a,i} = \sum_g CF_{g,i} \times p_{g,a} \quad \text{(Eq. 5)}$$

$$BD_{import,a,r,EU} = \sum_i A_{a,i,r} \times t_{a,EU,r} \times CF_{a,i} \quad \text{(Eq. 6)}$$

$$BD_{import,r,EU} = \sum_a BD_{import,a,r,EU} \quad \text{(Eq. 7)}$$

The allocation of $BD_{import,r,EU}$ to single EU member states was obtained by multiplying the damage by the share of net import for each country and product.

In case of a negative net import of products (i.e., a net export) from the EU region to the other regions, the amount *t* of product *r* exported from the EU was allocated to the member states *c* according to their share of net export for each product ($t_{c,r}$). The country specific characterization factors were obtained from the $CF_{g,i}$ multiplied by their area share $p_{g,c}$ as occupied by each ecoregion in the country *c* (Eq. 8). The exported amount $t_{c,r}$ was multiplied by the area demand $A_{c,i,r}$ of land use type *i* for producing one unit of product *r* and by the country specific characterization factors $CF_{c,i}$ to obtain the biodiversity damage due to net export $BD_{export,c,r}$ (Eq. 9).

$$CF_{c,i} = \sum_g CF_{g,i} \times p_{g,c} \quad \text{(Eq. 8)}$$

$$BD_{export,c,r} = \sum_i A_{c,i,r} \times t_{c,r} \times CF_{c,i} \quad \text{(Eq. 9)}$$

For each EU member state, the biodiversity damage due to net exports was deducted from the other damages in the computation of the EU biodiversity footprint.

The biodiversity damage due to internal trade within the EU28 region was also considered. For each member state, the net export/import amount for each product was converted into BD as in Eq. 6-9 and added to, or deducted from the impacts due to land use in each country.

2.4 Policy scenarios

In order to analyze the implications of increasing bioenergy consumption, three prospective scenarios were considered. The scenarios were developed to depict different pathways for the future development of the EU bioenergy sector (Forsell et al. 2016).

Baseline (BASE)

The Baseline scenario (BASE) was specified as close as possible to that of the EU Reference Scenario 2013 published by the European Commission. The Baseline scenario depicts the development of biomass use under bioenergy policies that aim at a 20% reduction of GHG emissions in the EU28 by 2020, but where the EU climate-energy targets for 2030 are not considered. The results show that increased demand for bioenergy will lead to a considerable increase in the EU domestic production of woody biomass (an increase by as much as 10% by 2030 in comparison to 2010 levels), as well as increased EU reliance on imported biomass feedstock, in particular wood pellet imports (an increase of 90% by 2030 in comparison to 2010 levels). From 2030 to 2050, the EU domestic production of biomass stabilizes as a result of slower development of EU bioenergy demand. The largest changes in the EU28 production of biomass feedstocks for bioenergy are seen in the development of SRC, which together with the EU import of wood pellets are expected to increase considerably in the future. In addition, there is an intensification in the use of EU forests, as well as an increase in the EU net import of roundwood. The increase in EU forest harvesting is driven by both the increasing demand for bioenergy, and the expected increase in demand of woody materials.

Constant demand

The Constant EU Bioenergy Demand scenario (CONST) investigates the effects of policies that increase the EU bioenergy demand similarly to the BASE scenario until 2020, but stay constant thereafter. There are only small differences between this scenario and BASE on the overall aggregate material production sector. However, compared to the BASE scenario, this scenario has more particleboard and less sawn wood production, driven by decreased demand for industrial by-products from sawmills (wood chips and sawdust) for bioenergy production. A clear difference is also seen in the composition of feedstocks used for energy production. Most importantly, the sourcing of domestically produced SRC and import of pellets is smaller than in the BASE scenario. Pellet imports increase until 2020, but remain almost constant thereafter.

Greenhouse gas emission reduction

The development seen in the BASE scenario is found to be accentuated in the EU Emission Reduction scenario (EMIRED), which builds on the policy target of decreasing GHG emissions by 80% by 2050 in the EU. In this scenario, the development of biomass use follows that of the BASE scenario until 2030. Thereafter, the results show a considerable increase in the EU import of wood pellets and domestic production of SRC. The increasing production of SRC in the EU after 2030 leads to some reductions in cropland and grazing land areas as compared to the BASE scenario, which in turn affects food and feed production. Additionally, we also see large quantities of roundwood directly used for bioenergy production in small and large-scale conversion facilities, especially by 2050. In other words, the bioenergy demand increases to an extent where stemwood that is of industrial roundwood quality and could be used for material purposes by the forest-based sector is instead being used directly for energy production. The increased use of biomass for energy has a direct impact on forest harvests, which are almost 9% higher than in the BASE results in 2050.

3. Results

3.1 Land use impacts occurring on EU territory

The area for the land uses considered in the EU from the year 2000 to 2050 increases by 12%, 10%, and 15% in the BASE, CONST, and EMIRED scenarios respectively (Figure 1 and SI 6 Table 1), as an effect of increasing bioenergy and food demand in the future.

The most relevant increase is the land area for wood extraction (forests and permanent crops), with a maximum in the EMIRED scenario, followed by the BASE and the CONST scenarios (for figures see SI 6 Table 1). Cropland expansion is relevant in the CONST and BASE scenarios, whereas it is less noticeable in EMIRED. This difference is due to the higher rate of conversion from cropland into perennial cultivations in EMIRED compared to the other two scenarios, especially after the year 2030 (Figure 1). Pasture land appears to be the category that is the least sensitive to the different bioenergy scenarios (Figure 1, SI 6 Table 1).

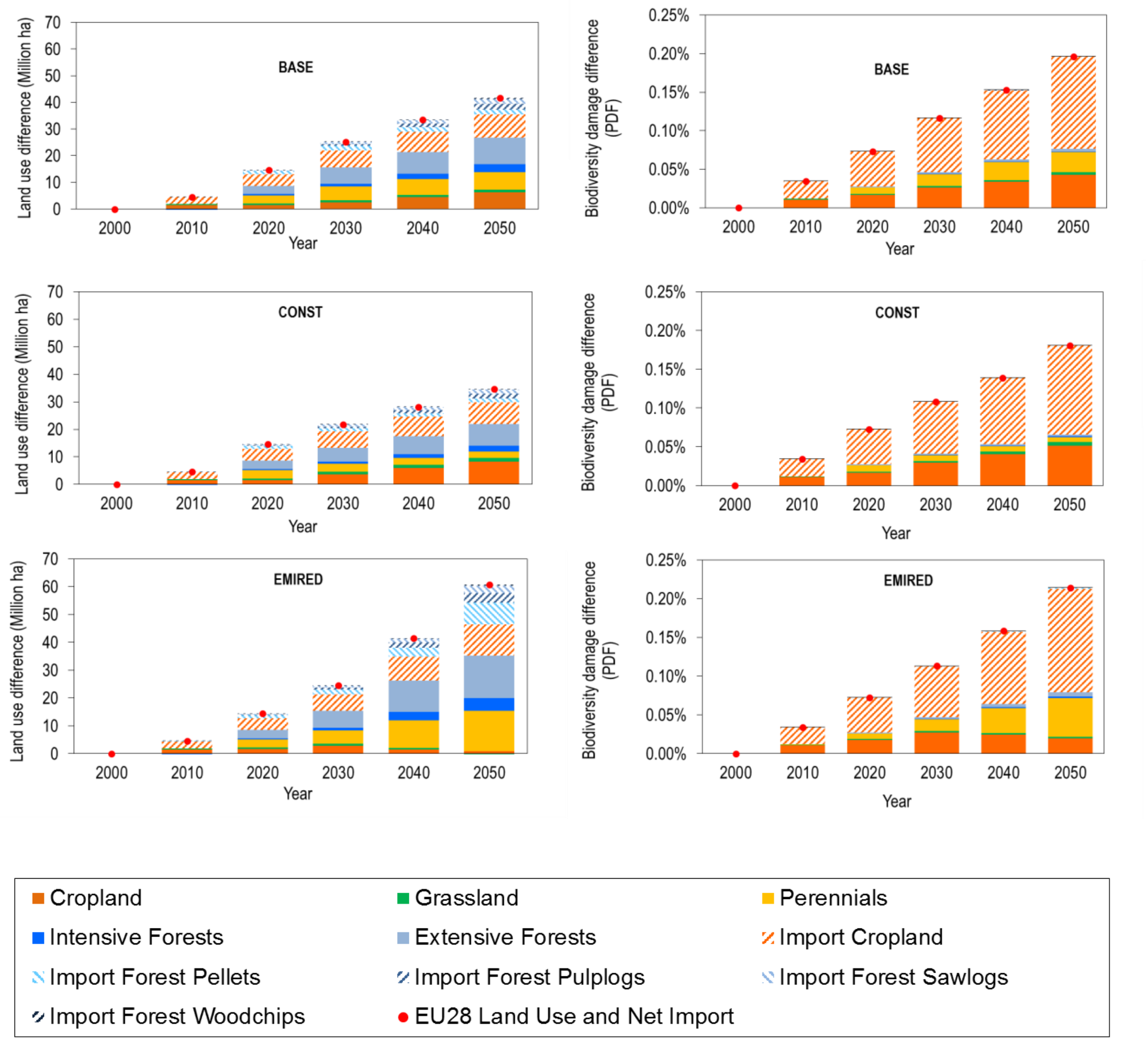
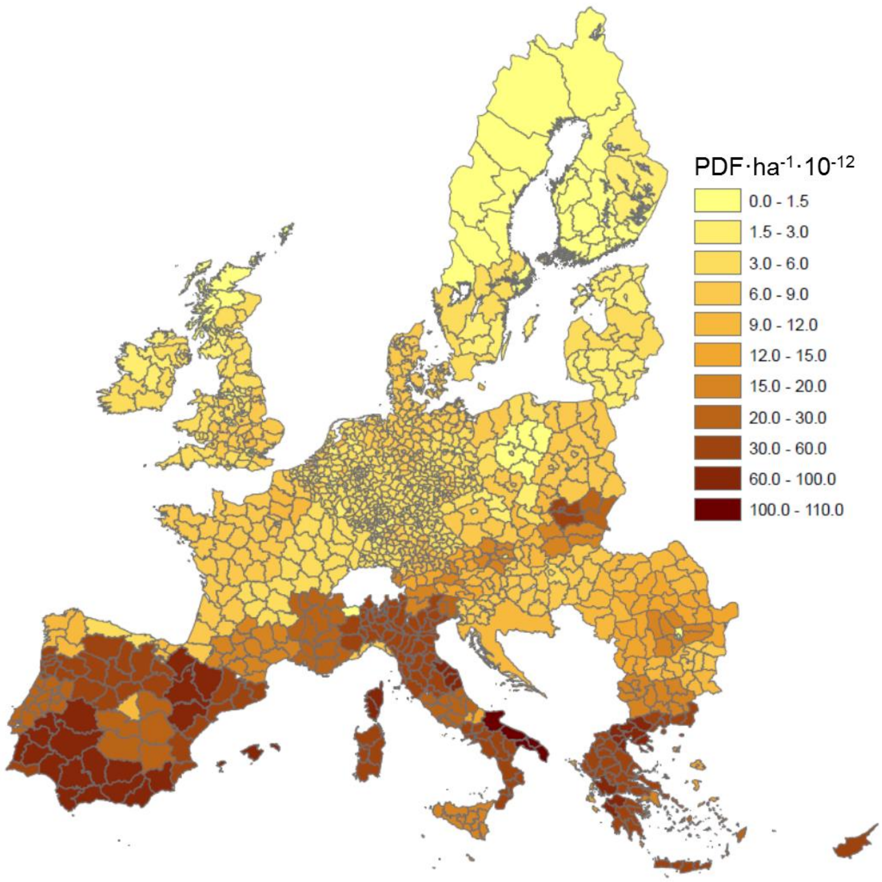


Figure 1. The graphs on the left show land use difference relative to the base year (2000) for the EU28 in the three different bioenergy scenariosas land uses within the EU, and net imported land uses. The graphs on the right depicts differences in biodiversity impacts relative to the base year (2000) due to land use within the EU and to net imports. The stacked columns represent the differences for each land use category compared to the year 2000, while the red dots represent the arithmetic sum of differences due to different land uses for each year.

The aggregated biodiversity impact due to land use in the EU28 from year 2000 to 2050 causes 0.08% of the global species extinction (7.63×10^{-4} PDF) in the BASE scenario. Cropland and grassland reduce their shares over time from respectively 78% and 16% (year 2000) to 76% and 15% (year 2050) of impacts. In the meantime, perennials reach 3.6% of land use impacts in the year 2050, while the share from used forests remains almost constant over time (6.0% to 5.9%).

The difference between the three scenarios increases after the year 2020: In the year 2050, the EMIRED scenario produces 0.4% more impacts than the BASE scenario, while the CONST scenario produces 1.5% less than the BASE scenario (SI 6 Table 2).

The impacts due to land use are amplified in South Europe, where the ecoregions are hosting more species richness than in the North (Figure 2-5). South western European countries show a total impact due to land use in the order of 0.1% of global species loss (10^{-3} PDF), while in the rest of EU countries the impacts are in the order of 0.00001% to 0.01% of global species loss (10^{-4} to 10^{-7} PDF). This spatial difference is magnified if considering the impacts per hectare of land (as PDF · ha, Figure 2).



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429 Figure 2. Biodiversity impacts in the units PDF·ha^{-1} for the BASE scenario in 2050 in the

430 EU28 NUTS2 administrative units due to land use (i.e., excluding trade).

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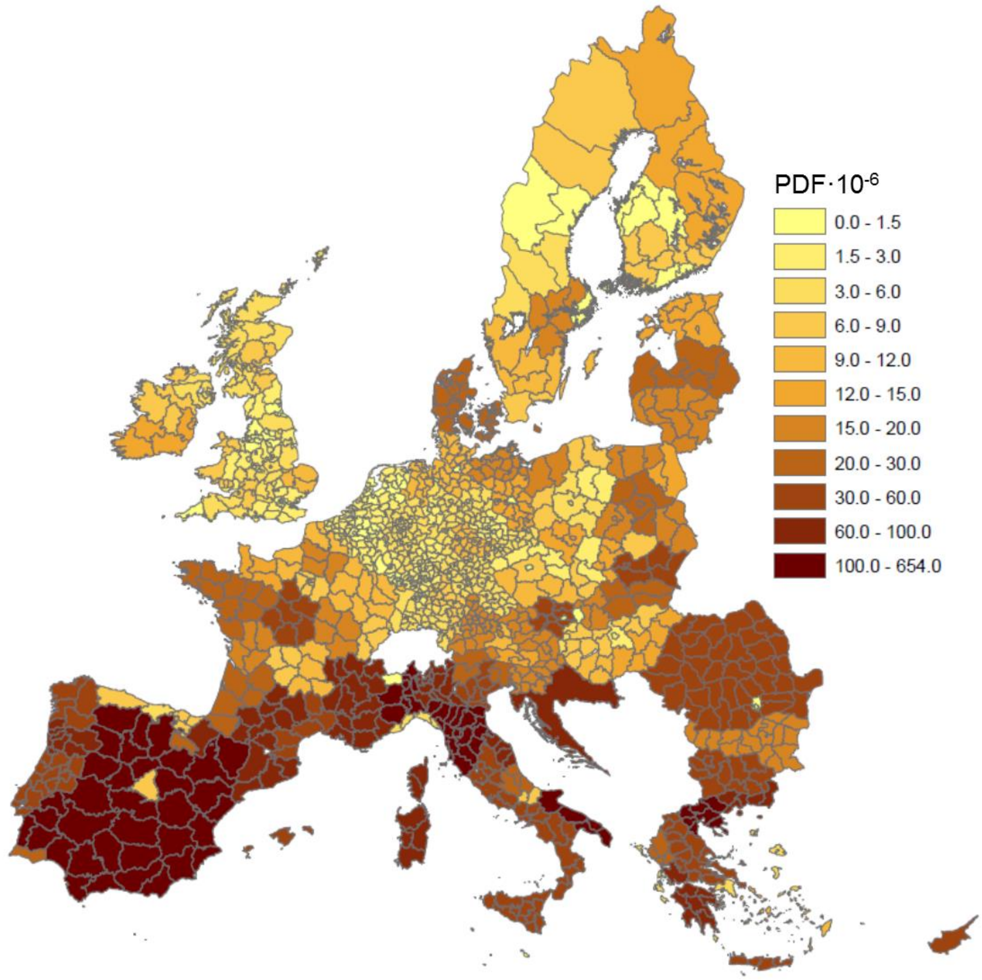

Figure 3. Biodiversity impacts in the units PDF for the BASE scenario in 2050 in the EU28 NUTS2 administrative units due to land use (i.e., excluding trade).

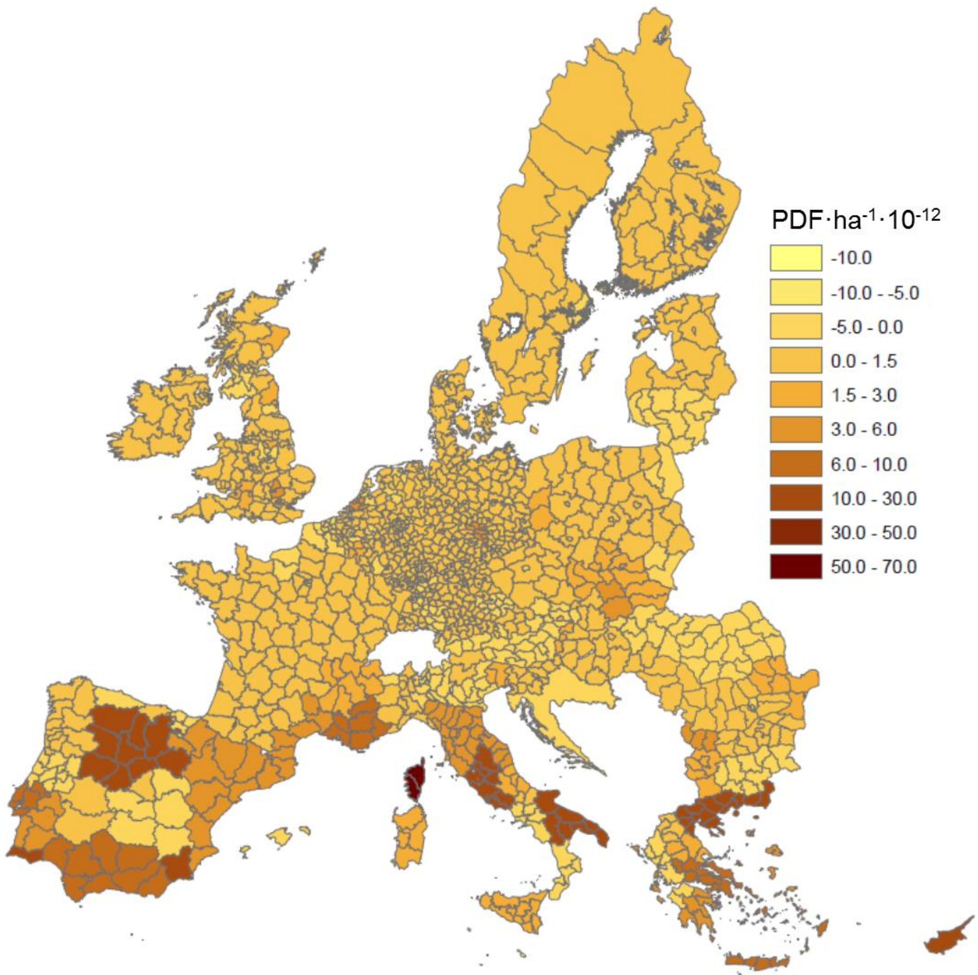

Figure 4. Biodiversity impacts in the units PDF·ha^{-1} for the BASE scenario as difference between the years 2050 and 2000 in the EU28 NUTS2 administrative units due to land use (i.e., excluding trade).

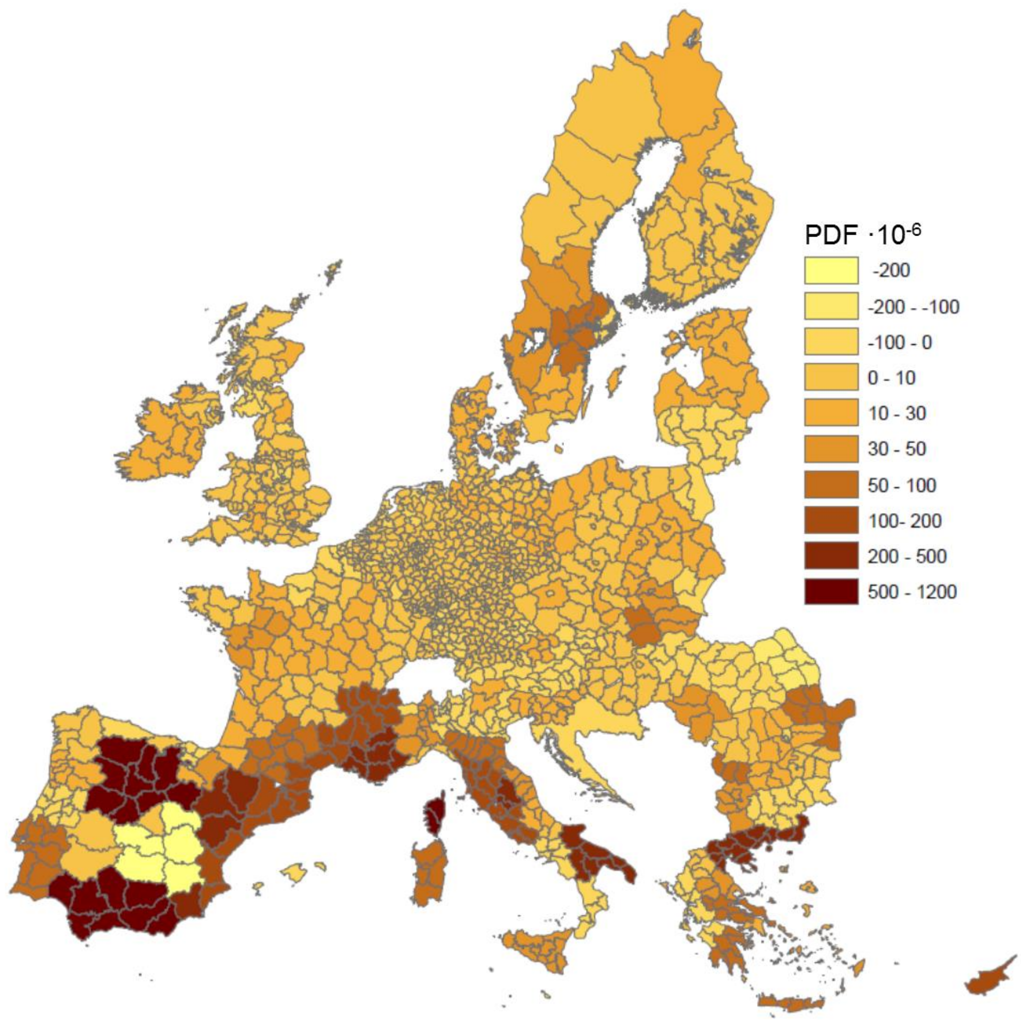

Figure 5. Biodiversity impacts in the units PDF for the BASE scenario as difference between the years 2050 and 2000 in the EU28 NUTS2 administrative units due to land use (i.e., excluding trade).

The biodiversity impacts from utilized forests increase in the future in all three scenarios due to a growth in roundwood extraction, which is expected to increase by 26% in the BASE scenario from the year 2000 to 2050. The corresponding numbers in the CONST and EMIRED scenarios are 20% and 41% respectively (SI 6 Table 3). Meanwhile, the surface of utilized forests increases by 19% in the BASE scenario, 15% in the CONST scenario, and 29% in the EMIRED scenario. The corresponding potential biodiversity damage due to forest management increases by 8.9%, 10.0%, and 18.5% over time in the CONST, BASE, and EMIRED scenarios respectively (SI 6 Table 3).

For all the scenarios, there is generally a reduction of impacts per unit of roundwood extracted over time, however the difference between the scenarios and over time is of a relatively small magnitude.

The impacts due to perennials (miscanthus and short rotation energy plantations) increase significantly over time in all scenarios; in the CONST scenario they stabilize after the year 2020, while in the BASE and EMIRED scenarios they continue to grow until the year 2050. In the year 2050, the potential biodiversity impact in the EMIRED scenario is almost doubled compared to the BASE scenario (i.e., a 95% increase) (Figure 6). The impacts due to the expansion of perennials is more relevant in the regions of South West and Central West Europe, representing 46% and 24% respectively of total damage in the EMIRED scenario in 2050 (Figure 6).

In the BASE scenario, the PDF per m^3 of perennials increases by 82% from 2010 to 2050, the corresponding increases are 22% and 48% in the CONST and EMIRED scenarios respectively (SI 6 Table 4). The impacts per m^3 in the CONST and EMIRED scenarios are similar to those of the BASE scenario from 2010 to 2030. After the year 2030, in these scenarios the impacts per m^3 are 11% to 33% lower than in the BASE scenario. This could be due to different reasons: in the CONST scenario, the demand for perennials is lower than in the BASE scenario, therefore the expansion of perennials is limited to natural vegetation and pasture land with relatively high yields compared to the land occupied by perennials in the BASE scenario. In the EMIRED scenario, the demand for perennials is higher than in the BASE scenario, which causes a further expansion of perennials in relatively high fertility croplands. However, in most of the regions, the increase of demand in the EMIRED scenario compared to the BASE scenario did not correspond to a significant expansion of perennials in croplands, leading to higher impacts per m^3 than in the BASE scenario.

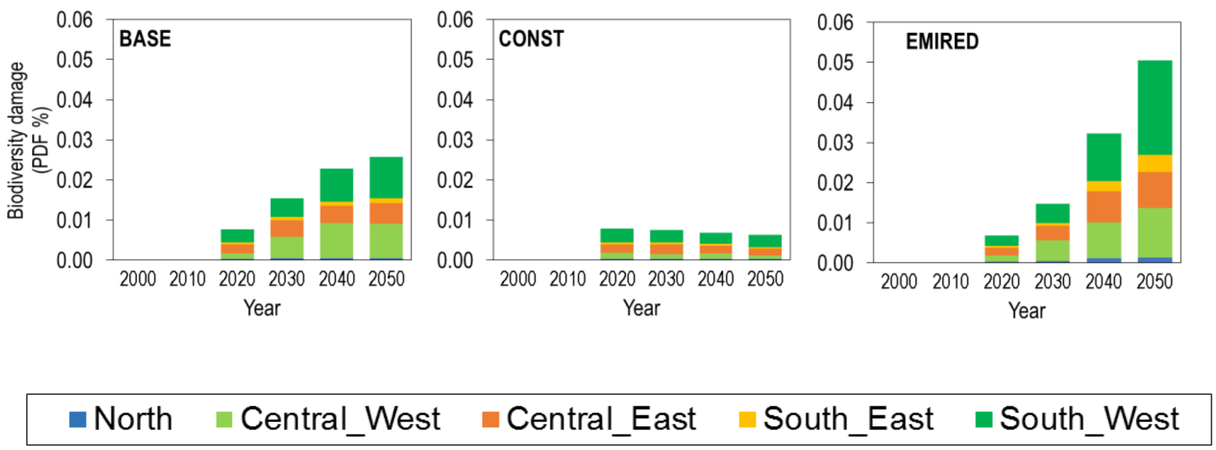

Figure 6: Development of biodiversity impacts due to land use in the units PDF from the year 2000 to 2050 due to perennial land use for the different regions of the EU (for a list of countries in each region see SI 5).

3.2 Impacts from trade

The net import of woody biomass from forestland to the EU28 progressively increases from the year 2000 to 2050. In the meantime, the EU28 increases the net import of cropland until 2030, and it then either stabilizes or continues to increase, depending on the scenario (Figure 1, SI 6 Table 5)In terms of traded product mass, the most important partners to the EU with regard to the net import of crops (i.e., from cropland) are Brazil, Australia, New Zealand, the Pacific Islands, Turkey, and Ukraine. From these regions, the most relevant imported crops are sugarcane, soy, rapeseed, sunflower, and cassava. Meanwhile, the most important net export regions are Africa and the Middle East. Our results show that over time, Canada and the former USSR also become relevant export regions. The most important net exported crops are wheat, corn, barley, and potatoes.

Pulp logs represent the largest share of net imports in terms of mass within the forest sector in the BASE and CONST scenarios, while in the EMIRED scenario, pellets achieve the same mass as pulp logs in 2040 and then in 2050 exceed pulp logs. The largest shares of pulp logs are imported from the former USSR and Malaysia, while for pellets the leading exporters are Canada, the former USSR, and the US (SI 6 Table 5).

The total biodiversity damage caused by net imports is in the order of 0.1% to 0.2% global species loss (1-2 × 10^{-3} PDF). Cropland causes 99% of this impact, and the remainder is mostly due to pulp logs (0.4-0.8%) and pellets (0.1-0.5 %) (Table 6). The impacts in 2050 are 2.2, 2.1, and 2.0 fold the ones observed in 2000 for the EMIRED, BASE, and CONST scenarios respectively. The differences between scenarios are amplified after the year 2030: In 2050, the impacts for the EMIRED scenario are 6% higher than for the BASE scenario, while for the CONST scenario they are 2% lower than in the BASE scenario (cf. SI 6 Table 1 and 6).

At the regional level, the largest shares of impacts due to net imports for the BASE scenario in the year 2000 are caused by Central West (54% of impact in the EU28) and South West Europe (39% of impact in the EU28). Over time there is a progressive increase of net imports for Central West Europe relative to the other regions (Figure 7). Consequently, Central West Europe causes 67% of EU28 damage due to net import in the BASE 2050 scenario.

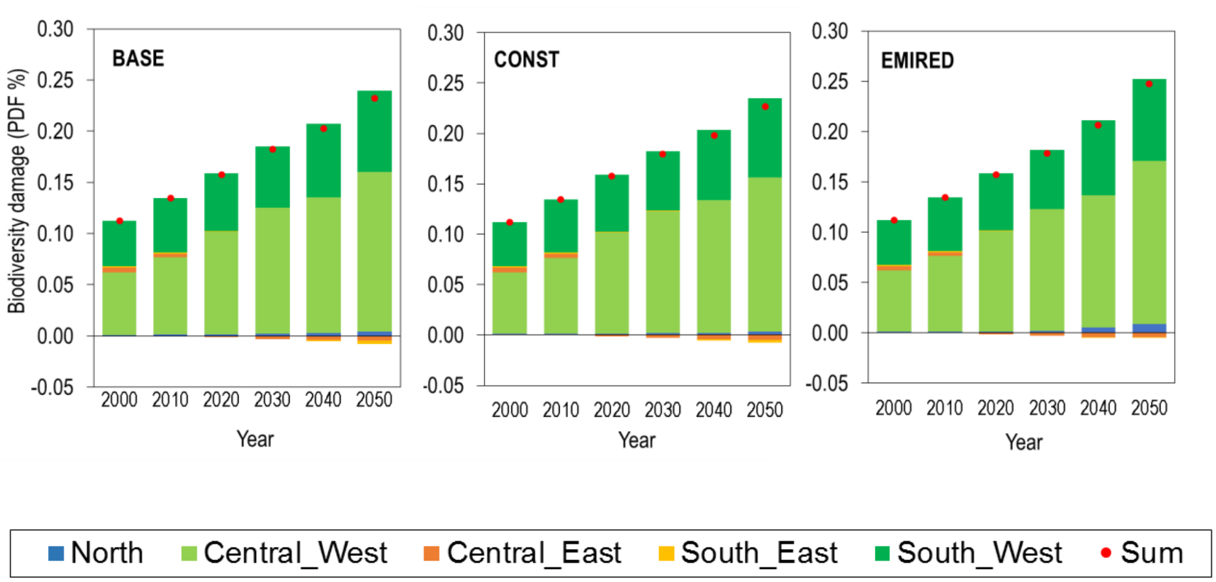

Figure 7: The development of biodiversity impacts due to trade in the units PDF from the year 2000 to 2050 expressed as the total of impacts due to external trade for the different regions within the EU regions. The red dots represent the arithmetic sum. Negative values denote net exports.

3.3 The biodiversity footprint of Europe

The total biodiversity damage, here referred to as the “EU footprint” was calculated as the sum of impacts due to domestic land uses in the EU28 summed to the impacts due to imports and decreased by the exports.

The EU footprint is in the order of 0.7% to 0.9% of global species loss (7-9 $\times 10^{-03}$ PDF). The impact of the BASE scenario increases by 26.1% from the year 2000 to 2050. The corresponding growths in the CONST and EMIRED scenarios are 24.1% and 28.6% respectively. The difference between scenarios is less than 1% until 2030. This increases over time and in the year 2050 impacts for the EMIRED scenario are 1.9% larger than in the BASE scenario. In the CONST scenario, they are 1.7% lower than in the BASE scenario (Table 1).

In all scenarios there is a growth over time in the share of impacts due to imports compared to land use, starting from 15% in the year 2000 and reaching 24% to 26% in 2050 (Figure 1).

After correcting for internal trade in the EU, the results show that countries in the Central West EU that are strongly dependent on imports (i.e., the UK) reach a total footprint (i.e., sum of land use and import) comparable in magnitude to the ones in the South West EU (cf. Fig. 2 and Fig. 4). The compensatory effect of imports is already evident in the year 2000 and increases over time.

In the BASE scenario, the countries with the largest share of the total footprint in the year 2000 are countries in the southwestern region. These countries represent 57% of EU impacts in the year 2000, and their share decreases to 54% by 2050 in the BASE scenario. Countries in Central West Europe, which are generally more dependent on net import, enlarge their share of the total EU footprint from 20% in 2000 to 26% in 2050. Similar tendencies are observed across all scenarios (Figure 8).

Table 1. Total biodiversity footprint from the EU (PDF), as the sum of impacts due to land use and net imports to the EU 28 in the three different bioenergy scenarios.

Year	2000	2010	2020	2030	2040	2050
Baseline (PDF)	7.50×10^{-03}	7.84×10^{-03}	8.23×10^{-03}	8.66×10^{-03}	9.02×10^{-03}	9.46×10^{-03}
Constant demand	0	0	-3.55×10^{-07}	-8.19×10^{-05}	-1.33×10^{-04}	-1.56×10^{-04}

(PDF Δ to baseline)						
Emission reduction						
(PDF Δ to baseline)	0	0	-7.08×10^{-06}	-3.32×10^{-05}	5.67×10^{-05}	1.79×10^{-04}

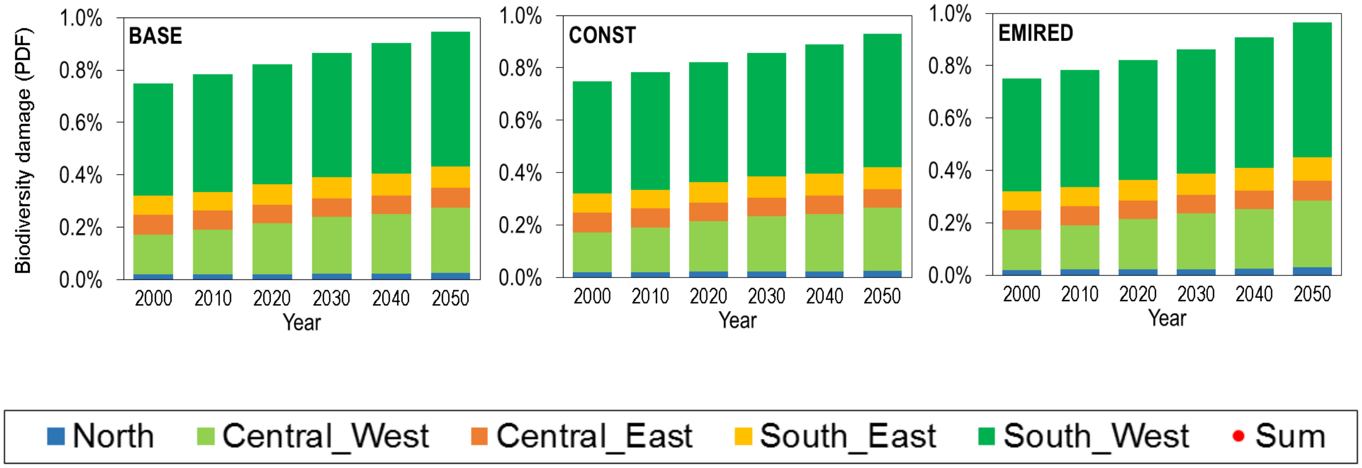

Figure 8: The development of the total biodiversity footprint from the EU in units PDF from the year 2000 to 2050 for the different EU regions.

4. Discussion

In this paper, we set out a global framework that is able to jointly assess and analyze the biodiversity implications of policies related to direct land use change, changes in intensity in land use and forestry, and in-direct land use effects. Utilizing this framework, we analyzed three different bioenergy policies in the EU28 and their effects on biodiversity, focusing on the expected changes in domestic land use and the possible damage on global biodiversity through trade.

In all scenarios, we observed a significant increase of biodiversity damage over time. In the long term (by 2050), the potential species loss due to the EU footprint was found to increase from

0.75% in 2000 to almost 1% of global species in 2050. Previous assessments suggest that ca. 10% of species globally could potentially have disappeared by 2050, compared to the year 2000 (CDB 2014). Given this background, the dynamics we analyzed for the EU28 have considerable impact on a global scale.

The increase of the biodiversity footprint over time is due to both an expansion of domestic land use and, especially, to land use imported through agricultural products into the EU. The international character of the problem is emphasized over time: The damage due to imported land use increased form 15% of total damage in the year 2000 to 24% to 26% in 2050, meaning that the footprint is progressively outsourced. This overall increase in the share of footprint caused by imports is mainly due to an increase of imported agricultural products to fulfill the growing European food demand and the area needed for this production outside the EU. This trend is reinforced by the conversion of cropland into perennials in the EU, which leads to outsourcing some of the cropland production to outside the EU. The biodiversity damage is magnified, as the imports of agricultural products include countries of origin located in tropical regions, in areas particularly rich of biodiversity and vulnerable species (i.e., Brazil, Australia, New Zealand, and the Pacific Islands). In these countries, the indirect damage per tonne of product is 5.9 to 8.9 times larger than in the EU. This result is in line with the findings of previous studies, which have found that the food consumption in industrialized countries drives biodiversity loss in tropical developing countries through international trade (Chaudhary & Kastner 2016, Lenzen et al. 2012).

Within the EU, agricultural production remains the largest domestic driver of land use related biodiversity impacts in all scenarios. The increase of food demand in the whole EU is expected to lead to a 1-8 Mha expansion of domestic cropland. However, the contribution of domestic cropland to the total EU biodiversity footprint (including imported land) is expected to decrease over time from 66% in 2000 to 54% to 59% in 2050. The most relevant future change of domestic land use in all scenarios is the expansion of perennial cultivation for energy, which is expected to increase to 2 to 14 Mha by 2050. The perennials are projected to increase especially after the year 2020, although their contribution to the total biodiversity footprint remains limited to 1% to 6% in 2050. Forests under active management in the EU expand over time by 10 to 20 Mha. Nevertheless, the domestic forest management area continues to be of minor relevance for the total biodiversity footprint (4% to 5% of the total footprint in 2050) compared to damages due to other domestic land uses and imported land use.

The difference between the three scenarios was found to be small compared to the magnitude of biodiversity damage increase over time. This finding is similar to the findings of Eggers et al. (2009), who also observed that different biofuel targets in the EU had a much smaller effect on biodiversity than the overall trend of biodiversity reduction observed over time from 2000 to 2030.

In our study, the scenario with the highest demand of bioenergy (EMIRED) created similar damage than the other two scenarios in 2050 (the difference between the scenarios was only 1.9% to 3.6%). In the EMIRED scenario, there is a larger expansion of perennials and a smaller expansion of cropland in the EU than in the other two scenarios. As the biodiversity damage is smaller in perennials than for cropland, this development lowers the internal biodiversity damage per unit of land occupied. In the meantime, in the EMIRED scenario there is also the largest outsourcing of damage, due to increased import of cropland products from outside the EU for satisfying the EU food demand. The two opposite effects even each other out, resulting in the total biodiversity damage for the EMIRED scenario being similar to the other two scenarios.

Over time, the growth in bioenergy demand also increases the import of wood pellets to the EU. In 2050, imports of pellets in the EMIRED scenario are 2.1 to 2.7 times higher than in the CONST or REF scenarios. However, the biodiversity damage created by wood pellet imports has only marginal relevance compared to the import of agricultural products, given the relatively lower characterization factors for managed forests compared to cropland and the main countries of origin for wood exports (USSR, Canada, and the US). In these countries, the biodiversity damage per unit of wood pellet is 4 to 24 smaller than the damage in the EU resulting from perennial cultivation.

The internal distribution of the EU footprint is determined by the split of the area into different land uses in each region in terms of the biodiversity richness in the different ecoregions, and especially by the amount of net imports. For these reasons, the largest biodiversity footprints in the EU were initially observed in the Mediterranean region, which is the region that hosts most of the biodiversity in the EU. However, over time, the biodiversity footprint increases in central-western EU countries that are particularly dependent on imports, due to the relatively more severe damage per unit of land caused by imports compared to the damage caused by domestic production of biomass (cf. Fig. 8).

We used only one indicator of species loss (potentially disappeared fraction of global species, PDF), which was obtained by aggregating the richness of species across the different taxa. A single indicator will not capture damages due to changes in species composition that take place following disturbances. The same methodology can be repeated through the use of characterization factors for the single taxa (cf. Chaudhary et al. 2015). Eggers et al. (2009) investigated the suitability of different species and concluded that mammals and birds were the most damaged by the expansion of biofuel crops. Therefore, to investigate the biodiversity damage in more detail, an investigation into impacts across the different taxa could be a possible extension of the current study. Furthermore, in the current study only global extinction-equivalents were accounted for, thus neglecting regional extinctions. However, the latter may also be important to warrant local ecosystem functioning and should be assessed in future research.

The strength of the approach that we proposed for evaluation of the biodiversity damage is that it is already consolidated in the literature (see UNEP/SETAC 2016). The biological functionality of an ecosystem, such as functional diversity, could not be assessed using species richness as an indicator. This is a topic for future studies, as there is currently a lack of available approaches for combining and evaluating different indicators of functionality (Maia De Souza et al. 2014).

In our calculations we assumed a steady state change in species extinctions and neglected the temporal evolution of biodiversity loss. In reality, the species will not instantaneously go extinct or return when land use change takes place. This delay in the species dynamics means that it is likely that the changes assessed in this paper will happen more gradually than assumed. We also did not consider aspects such as land fragmentation or ecological corridors, which are important to biodiversity with regard to landscape level continuity. Landscape analyses could help to understand the effects of different patterns of land use.

The economic model used in this study produced land use projections at the resolution of NUTS2 administrative units. Trade was modeled between global trade regions, and internal trade within the EU was modeled at the country level. The characterization factors were originally available on an ecoregion scale, hence they were re-scaled to fit the different resolutions (NUTS2, Country, trade region). Although some accuracy is lost through this re-scaling, the geographical scale is still rather small and is considered to be a strength of the study.

In our study we estimated the intensity of forest management in the EU using a suitability map, which considered 28% of managed forests under intensive management.

Intensifying the use of forest biomass could also affect forest management regimes, leading to a reduction in rotation periods, a possible increase of monocultures, or collection of residual wood debris. These could in turn negatively affect biological diversity and natural habitats, which could lead to further reductions of local biodiversity (Lassauce et al. 2012, Söderberg & Eckerberg 2013). All these aspects could result in more substantial damage to the hosted biodiversity compared to our analyses.

Using the growth rates for plantations predicted in ABARE & Pöyry (1999) and Jürgensen et al. (2014), which assumes further intensification of European forest management than what resulted from our scenarios, intensified managed forests in the EU could reach 39% of managed forests in 2030 and 55% by 2040-2050. In this new condition, the potential species loss in 2050 would increase and forests would cause 7.4% to 7.8% of the internal land use damage. However, we considered only two classes of intensity, while Chaudhary et al. (2016) provided response ratios for then different classes of forest management. They found significant and different species losses produced by plantations, clear-cutting, and conventional selective logging. This suggests that forest management intensity may have a larger effect than what is shown in the current study. Our simplification was due to the scarcity of data regarding forest management statistics, which did not allow us to distinguish globally among more than the two classes.

The intensity of forest management outside the EU was based on the regional statistics of wood supply from planted forests reported in Jürgensen et al. (2014) and projected according to the long term growth rates estimated by ABARE & Pöyry (1999). Currently, there is a lack of data for validating the area of forest plantations. The statistics that are globally available from the Food and Agriculture Organization of the United Nations (FAO)'s Forest Resource Assessment (FRA 2015), report the surface and change rate of "planted forests" per country without specifying the different uses of the planted forests (production, protection, etc.). In some regions, the current growth rates observed in FRA 2015 could deviate from the ones projected in our study. However, it is not straightforward to distinguish planted forests from plantations that currently contribute to wood supply in the different regions. The current EU import of forest biomass is mostly sourced from forests and wood plantations assimilated to intensively managed forests. For this reason, we assumed future wood imports to have also originated from intensively managed forestland. If considering the most biodiversity adverse situation (the whole EU import of wood pellets in 2050 will be sourced from perennial plantations), the damage due to imports of wood products in 2050 would increase by a factor of 1.2 to 1.5. Under this condition, the biodiversity damage per m^3 of wood pellets imported is still 2 to 17 smaller than the damage for perennials in the EU. However, more significant damage could be induced if perennial plantations outside the EU would displace food production. If considering both the domestic and external intensification in the supply of woody biomass, the potential species loss in 2050 would increase by a factor of 1.01.

The GLOBIOM model used for projecting land uses is based on the economic convenience of allocating land to different uses, or in practical terms, on demand and supply curves. The growth of a bioeconomy in the EU could lead to an intensification of local demand points in some regions where the industry could expand more easily, and this could reduce costs and increase the profitability of supplying biomass locally (Hellman & Verburg 2011). This development could significantly alter the land use allocated to biomass production within the different regions compared to our results. Therefore, the results must be seen as representative of general trends in the EU, and not as being an exhaustive description of development within each administrative unit.

Agricultural and forestry yields were kept constant in our study, which could have led to an overestimation of damages produced by future cropland expansion. Increasing agricultural yields in regions with significant yield gaps could lead to intensification, which could further lead to future sparing of land from agriculture and instead utilizing it for possible bioenergy use (Lamb et al. 2016). Consequently, we were rather conservative from this point of view.

5. Conclusions

Our results show that policies promoting bioenergy in the EU may contribute to a further global decline of biodiversity. While a strong expansion of perennial crops for bioenergy production could be an interesting option for climate change mitigation, it could have negative impacts on biodiversity through loss of species habitats. Further, our results indicate that through international trade, an increase in bioenergy demand may result in a considerable leakage of biodiversity damage to species-rich and vulnerable regions outside the EU. Therefore, in the case of future increase in bioenergy demand, the combination of supply from sustainable forest management in the EU and imported wood pellets combined with the cultivation of perennial energy crops, appears to be less detrimental to biodiversity than only an expansion of energy crops.

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