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A comparison of land use change accounting methods: seeking common grounds for key modeling choices in biofuel assessments

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Abstract

Five currently applied methods to account for the global warming (GW) impact of the land-use change (LUC) induced greenhouse gas (GHG) emissions have been applied to four biofuel case studies. Two of the investigated methods attempt to avoid the need of considering a definite occupation –thus amortization– period by considering ongoing LUC trends as a dynamic baseline. This leads to account for a small fraction (0.8%) of the related emissions from the assessed LUC, thus their validity is disputed. The comparison of methods and contrasting case studies illustrated the need of clearly distinguishing between the different time horizons involved in life cycle assessments (LCA) of land-demanding products like biofuels. Absent in ISO standards, and giving rise to several confusions, definitions for the following time horizons have been proposed: technological scope, inventory model, impact characterization, amortization/occupation, plantation lifetime and harvesting frequency. It is suggested that the anticipated technical lifetime of biorefineries using energy crops as feedstock stands as the best proxy for the cut-off criterion of land's occupation period, and

thus for the amortization and inventory modeling periods. Top-down LUC models are suggested as a gross reference benchmark to judge LUC results from bottom-up models, since the former represent average GHG emissions from deforestation statistics at different spatial resolutions. Reporting LUC emissions per area and implementing a corporate accounting system that ascribes deforestation emissions to responsible companies could avoid the critical uncertainty related to yield estimations.

Keywords: land use changes, biofuels, life cycle assessment, time horizons, dynamic baseline method, reference system for LUC.

Highlights

- The technical lifetime of biorefineries is suggested as a technical and empirical metric for determining the long-term occupation of the used land. This, in turn, can be the basis to determine the amortization (if applied) and inventory modeling periods.
- The validity of current dynamic baseline methods to account for GHG emissions from LUC is disputed.
- Top-down LUC models represent average GHG emissions from deforestation which can be used as a gross reference to assess the validity of bottom-up LUC model results.

1. Introduction

After the European Union (EU) and the United States (US) established legal requirements for minimum biofuel use several years ago (European Commission, 2009; U.S. Congress, 2005), land-use change (LUC) emission accounting at the product level opened up the biofuels' box of Pandora (Fargione et al., 2008; Searchinger et al., 2008). The guidelines of the Intergovernmental Panel on Climate Change (IPCC) for national greenhouse gas (GHG) inventories (IPCC, 2006) were initially maladapted by several policies, as biogenic LUC emission were erroneously considered to be carbon neutral (Searchinger et al., 2009). The importance of this error was later reiterated and its implications for GHG accounting studies highlighted (Haberl et al., 2012). On the premise that only bioenergy from “additional biomass” can reduce GHG

emissions, the use of food crops or arable land for bioenergy has been repeatedly questioned. This is because land-demanding bioenergy may induce the displacement of the previous agricultural production elsewhere (Haberl et al., 2012). Assuming that food (and generally land) demand will not decrease in the future, energy cropping ultimately leads to land expansion and intensification, i.e. indirect LUC (iLUC), inducing significant GHG emissions (Kløverpris, 2008; Schmidt et al., 2015). This leakage or iLUC effect has been addressed by many studies that focus on different biofuels and crop feedstock in different regions, like the ones assessing the US and the EU's biofuel policy implications (U.S. Environmental Protection Agency (EPA), 2010; Valin et al., 2015). The considered price-demand elasticities (i.e. how demand changes with price changes), future yields and other necessary modeling assumptions result in a wide range of iLUC emission factors (Plevin et al., 2015; Smith et al., 2014), which makes their implementation in policy-making difficult or even controversial (Finkbeiner, 2014). Several studies have proven the dominant role of LUC in determining the environmental performance of biofuels from a life-cycle perspective, thus their inherent uncertainty should not be a reason to exclude them in environmental assessments (Muñoz et al., 2014; Plevin et al., 2010; Sanchez et al., 2012). Reported uncertainties rather point to an urgent need to further examine the key parameters that determine iLUC emissions (Plevin et al., 2015) to reduce results uncertainty (Li et al., 2016) and create consensus on key assumptions.

Life cycle assessment (LCA) is a standardized environmental assessment methodology (ISO, 2006a, 2006b), well-acknowledged and increasingly used to assess the environmental impact of biofuels and biofuel policies worldwide (U.S. EPA, 2010). One of the most critical key assumptions in LCA studies of land-based products is the amortization period used for LUC (Plevin et al., 2015), i.e. the time horizon over which the LUC emissions are linearly distributed for accounting. This is an artificial construct, which does not reflect the real GHG emission dynamics. Most policies regulating life-cycle GHG emissions of land-based products today recommend a 20 years period for amortization as recommended by the IPCC guidelines (BSI, 2011; European Commission, 2015; Greenhouse Gas Protocol, 2011; ISO, 2013). Since life-cycle results are typically given per functional output (e.g. per MJ for biofuel studies), reported LUC emissions (and the respective results) vary dramatically with the assumed amortization period (Kløverpris and Mueller, 2012)

72 and estimated crop yields (Plevin et al., 2015). As an alternative, and to avoid the arbitrariness of a fixed
73 amortization period, some life-cycle approaches have suggested taking a dynamic land-use baseline and
74 proposed “amortization-free” LUC emission factors (Kløverpris and Mueller, 2012; Schmidt et al., 2015).
75 Despite these significant efforts to model the environmental consequences from demanding additional land
76 all over the world for growing various feedstocks (agricultural expansion in particular, but also
77 intensification as in Tonini et al., 2016), there is a lack of research to identify the differences between the
78 existing LUC accounting methods. Likewise, a need to distinguish the different time horizons involved in the
79 LCA of land-demanding products has been identified.

80 Even though LUC are relevant for any land-demanding product, this study focuses on biofuels as they have
81 received most of the scientific attention for decision-support in policy-making (European Commission, 2015;
82 Haberl et al., 2010; U.S. EPA, 2010). Four different biofuel case studies have been selected to illustrate the
83 induced LUC emission estimates that result from applying five different LUC accounting methods. Their
84 differences are presented and the possibility of a common ground for the key amortization period assumption
85 is investigated. The appropriateness of some methods is critically discussed, as well as the confusion around
86 different time horizons in LCA and their relationship to biofuel assessments. The article concludes with the
87 relevance of the analyzed methods for future LUC modeling and some potential policy implications.

88 **2. Methods**

89 **2.1 Time horizon definitions and LCA principles**

90 The ISO (ISO, 2006a) standards for LCA are based on the polluter-pays principle (Schwartz, 2005). This
91 states that polluters are responsible for the environmental impairments they generate and hence, they need to
92 compensate for the costs derived from reversing the harm. For LCA and environmental footprinting methods
93 that take a life-cycle perspective, this translates into the obligation of accounting for all the emissions caused
94 by the product along its entire life cycle. As a consequence, emission accounting methods such as the
95 analyzed LUC models should strive for complying with the completeness, transparency, relevance and
96 accuracy principles (Greenhouse Gas Protocol, 2011; ISO, 2006b).

97 On the other hand, the ISO standards for LCA do not specifically define the different time horizons involved
98 in a LCA. Therefore, some definition proposals are introduced here, as a prerequisite to advance in the
99 conundrum around time horizons:

- 100 - **Technological time scope:** This time horizon is related to the life-cycle of the assessed product. It is
101 part of the system boundary definition in the goal and scope phase (ISO, 2006a). When the assessed
102 product or service requires the direct implementation of a producing technology (in the foreground
103 system, e.g. a biorefinery plant), the technological time scope refers to the minimum anticipated
104 lifetime of the technology over which the service or product is delivered.
- 105 - **Inventory modeling period:** This is the time horizon over which emissions are accounted, also
106 referred to as ‘analytical horizon’ (Sanchez et al., 2012). In most cases, the inventory modeling
107 period will end when the product reaches its end of life and is disposed of (and thus coinciding with
108 the technological time scope). In some cases, long-term emissions may be expected, e.g. metal
109 leaching from landfill (Hauschild et al., 2008) or LUC-induced peat oxidation (Valin et al., 2015).
110 Such long-term emissions must be included within the system boundaries to comply with the
111 completeness and accuracy accounting principles. For this, an extended inventory modeling period
112 may be required (Bakas et al., 2015; Hauschild et al., 2008) which goes beyond the life cycle of the
113 assessed product, and hence beyond the technological time scope of the LCA.
- 114 - **Impact modeling period:** This is the time horizon used by the impact assessment methods in the
115 impact characterization step (ISO, 2006a); e.g. the time horizon used to calculate the global warming
116 potential (GWP) of different GHG emissions. Due to the sensitivity of the impact score to the
117 modeling period used in its characterization, it needs to be clearly stated for reporting purposes (ISO,
118 2006a). For example, if a 100 years horizon is used to report global warming (GW) impacts, the
119 impact assessment method would be reported as GWP₁₀₀.

120 In addition to the these time horizons that apply to any LCA, there are three other relevant periods for biofuel
121 assessments (or other land-dependent products) that are involved:

- **Amortization period:** Borrowed from financial accounting, the amortization period represents the assumed time horizon over which the assessed activity will take place and thus, the period over which the (environmental) investments need to be distributed (and possibly paid back). For land-intensive products like dedicated biofuels, it refers to the period over which land is expected to be occupied for the production of the raw material.
- **Harvesting frequency or single-rotation:** refers to the time period between two consecutive harvests.
- **Plantation lifetime or full-rotation:** refers to the period between the planting and the final removal of a perennial crop plantation.

2.1.1 Harmonization of applied time horizons

While the impact modeling period to report GW impacts is rather homogeneous in literature (100 years by convention), the technological time scope is generally ignored and the inventory and amortization periods vary according to the case. In order to facilitate the comparison of different methods, the time horizons need to be harmonized based on the given definitions listed above

Energy crops like corn or sugarcane will be indeed replanted as long as they are demanded by fermentation plants to produce and supply bioethanol and/or bio-based products. Biorefineries and bio-based power-plants are necessarily inside the system boundaries of biofuel and bioenergy LCA, hence their technical lifetime stands as a more reasonable, robust and relevant criterion than any other arbitrary amortization period choice (European Commission, 2009). Despite still subject to the inherent political and economic uncertainties around the long-term operation of a plant, this criterion is commonly applied in industry to derive the economic viability of an investment. Albeit some uncertainty, the technical lifetime is based on material science, i.e. endurance of the components and materials, and empirical data from industries.

For the sake of simplicity, it was assumed that no peat land has been drained for the establishment of any plantation. This means that the technical criterion that determines the occupation period of land (and thus the amortization period) is also valid for determining the GHG inventory modeling period, since no long-term

emissions are expected for these case studies. These and other aspects around long-term emissions are discussed in Section 4.1.1.

Table 1. Time horizons in the four study cases, according to the definitions and criteria presented in Section 2.1. All values are given in years.

Biofuel study cases	Single rotation	Full rotation	Technological scope	Amortization ^o (occupation)	Inventory modeling ^o (GHG emissions)	Impact modeling (GWP)
Willow woodchips	3	20	20	20	20	100
Sugarcane ethanol	1	6	30	30	30	100
Palm-oil biodiesel	-	25	30	30	30	100
Corn ethanol	1	1	30	30	30	100

^oBoth the amortization and inventory modeling periods represent the expected occupation period, which was selected according to the presented criterion (the technological scope).

2.2 Biofuel study cases

The used land area has been taken as the functional unit (FU) for this assessment (Cherubini et al., 2009; Pawelzik et al., 2013). Reporting the LUC per demanded hectare (ha_{dem}^{-1}) allows for estimating the resulting GHG emissions independently of the related crop yields, thus avoiding the propagation of the inherent variability and uncertainty of yield estimates (Plevin et al., 2015) to the GHG emission estimates from LUC. To cover a wide range of biofuel types, four energy crops from different regions of the world and with different plantation life-cycles were selected: oil-palm, short-rotation willow, sugarcane and corn (see Table 1). It was assumed that the oil-palm plantation was used for biodiesel and established on a Malaysian logged-over forestland (Wicke et al., 2008), the willow was gasified for cogeneration of heat and power (CHP) and established on Danish arable land (Saez de Bikuña et al., 2016), while the sugarcane and corn were used for ethanol production and were planted on Brazilian cropland (Lapola et al., 2010) and on US cropland (Plevin et al., 2010), respectively. The single- and full-rotation periods for Brazilian sugarcane were taken from Seabra et al. 2011. Following the criterion presented in 2.1.1, the most relevant occupation (i.e. amortization) periods were identified for each biofuel case: 30 years for the ethanol and biodiesel feedstock (as that is the expected lifetime of such biorefineries; Davis et al., 2013; U.S. EPA, 2010) and 20 years for willow CHP (both the lifetime of the plantation and that of a small cogeneration plant, Energinet, 2012; Saez de Bikuña et al., 2016).

170 2.1.1 *Ad hoc* LUC emission factors

171 For comparison purposes (and since iLUC emissions cannot be directly measured), we take as reference
172 value the LUC emissions estimated *ad hoc* by the authors (Lapola et al., 2010; Plevin et al., 2010; Saez de
173 Bikuña et al., 2016; Wicke et al., 2008). The *ad hoc* LUC emissions were estimated with case-specific
174 economic iLUC models (bottom-up) based on consequential LCA. These models predict the multiple
175 economic effects related to supply-demand laws, such as product and co-products substitution,
176 intensification and reduced (food/feed) demand, in order to get a final area expansion estimate from the
177 initial demand shock. The LUC emission of the Malaysian palm-oil biodiesel represents only direct LUC,
178 because the land clearing is assumed to take place directly on the native rainforest to establish the oil-palm
179 plantation (Wicke et al., 2008).

180 For the Danish willow plantation, LUC emissions were taken from Table 2 (iLUC emissions) and soil C
181 gains of $28.5 \text{ kg C ha}^{-1}\text{yr}^{-1}$ are considered (Methods section in the main text) based on Saez de Bikuña et al.,
182 2016. For the Brazilian sugarcane plantation, LUC emissions were derived from the total carbon debt of
183 sugarcane expansion (from 4224 Tg CO₂eq and 13.6 Mha expansion, Figure 2, Lapola et al. 2010), which
184 yielded a LUC emission factor of $311 \text{ Mg CO}_2\text{eq ha}_{\text{dem}}^{-1}$. The LUC factor for corn ethanol was taken as the
185 mean value of several studies ($62 \text{ g CO}_2 \text{ MJ}^{-1}$) and the average ethanol yield of $4000 \text{ liter ha}^{-1}$ (Plevin et al.,
186 2010). Taking a lower heating value (LHV) of $22.8 \text{ MJ liter}^{-1}$ (Seabra et al., 2011), resulted in $169 \text{ Mg CO}_2\text{eq}$
187 ha^{-1} . For the Malaysian oil-palm plantation, LUC emissions were calculated from the C stock data provided
188 in Table 2 in Wicke et al. 2008. A logged-over rainforest with an above-ground vegetation of 175 Mg DM
189 ha^{-1} was considered, which corresponds to 88 Mg C ha^{-1} . To this, a loss of 20 Mg C ha^{-1} from the soil was
190 added (Table 2, Wicke et al. 2008), which resulted in a total LUC estimate of $388 \text{ Mg CO}_2 \text{ ha}^{-1}$.

191 Temporary plant C-sequestration was neglected for corn, sugarcane (annually harvested) and willow (3-year
192 harvests), while considering it for the oil-palm tree plantation with a GWP_{bio} factor (Cherubini et al.,
193 2011). Therefore, the permanent plant sequestration of 95 Mg C ha^{-1} considered by Wicke et al. 2008 was
194 converted to temporary C-sequestration with a GWP_{bio} factor of 0.1 for a 100 years horizon and 26 years

195 rotation period (Table 3 in Cherubini et al. 2011). Table 2 summarizes the values and key characteristics of
 196 the *ad hoc* LUC emission factors.

197 Table 2 *Ad hoc* LUC emission factors, their main characteristics and values.

Energy crop	Model type	Value (Mg CO ₂ eq ha _{dem} ⁻¹)	Reference
Willow	Bottom-up. GTAP model	283	Saez de Bikuña et al., 2016
Sugarcane	Bottom-up. LandSHIFT model	311	Lapola et al. 2010
Oil-palm	Direct LUC	388	Wicke et al. 2008
Corn	Mean of several bottom-up models	162	Plevin et al. 2010

198

199 2.3 LUC accounting methods

200 Five different LUC accounting methods are taken to illustrate the variation of accounted LUC for each
 201 biofuel case per additional hectare demanded. The five accounting methods are explained in the following
 202 subsections and comprise of: *i*) a global average LUC emission factor that includes both expansion and
 203 intensification effects, LUC_{global}; two versions of a dynamic land-use baseline method: *ii*) LUC_{DBM1} and *iii*)
 204 LUC_{DBM2}; *iv*) the consequential variant of the GHG protocol method for accounting unknown LUC of
 205 products, LUC_{GHGP}; and *v*) the iLUC emission estimates computed for the last amendment to the European
 206 Renewable Energy Directive (RED), LUC_{RED15}. The dynamic baseline methods deal also with the issue of
 207 amortization and, indirectly, with the conundrum around time horizons (Kløverpris and Mueller, 2013). All
 208 of the methods lead to results that can be expressed in mass CO₂eq emitted per hectare for all of the
 209 investigated cases.

210 2.3.1 World-average LUC emissions: LUC_{global} factor

211 This top-down method, also called biophysical (Schmidt et al., 2015) or deterministic (Tonini et al., 2016;
 212 Warner et al., 2014), is used herein to calculate a generic LUC_{global} factor. This is based on previous work
 213 (Saez de Bikuña et al., 2016) and is fully described in the Supplementary Materials (SM) in Appendix A.
 214 This method provides a global average LUC emission factor representing the approximate effect of
 215 demanding one additional productive hectare to the global market. It is assumed that the land appropriation
 216 for energy cropping displaces certain food crop production, which is considered to be achieved through a
 217 combination of agricultural expansion and intensification processes. The share of each process is computed

218 on the basis of time series analysis from global food production statistics (FAO stat database). The additional
 219 food production is thus achieved through agricultural land expansion and additional synthetic nitrogen
 220 fertilizer application in the respective (historical) proportions, thereby assuming linearity for future trends.
 221 Here, it is considered that 37% of the additional global food production is achieved through land expansion
 222 and 63% through intensification (Saez de Bikuña et al., 2016), calculated on a wet basis (differently from the
 223 dry basis shares of 25/75% as taken in Tonini et al., 2016). Contrary to economic LUC models, other short-
 224 term indirect effects like reduced food consumption (Valin et al., 2015) are disregarded, as it is assumed that
 225 the mid- and long-term food demand is not affected by short-term price changes (Schmidt et al., 2015). Since
 226 biofuel production periods (i.e. the explained occupation and amortization periods) will span over two or
 227 three decades (see Table 1), this is a powerful simplifying assumption which is considered to be valid under
 228 the logic of economic supply-demand laws. That is, as long as the production factors (for the price affected
 229 food/feed crops) are not constrained (and, in the light of development and within a productivist view, land is
 230 not yet a constraint), supply will follow demand insofar market price pays off production costs. As a result,
 231 the initial food shock demand is assumed to be fully satisfied in the mid- or long-term, i.e. within the
 232 duration of the assessed bioenergy systems. The resulting average GHG emissions included in the LUC_{global}
 233 factor are therefore $165.5 \text{ Mg CO}_2\text{eq ha}_{dem}^{-1}$ (from agricultural expansion, which are amortized according to
 234 the suggested criteria, over the stated occupation periods; see 2.2) and $2.1 \text{ Mg CO}_2\text{eq ha}_{dem}^{-1} \text{ yr}^{-1}$ (from
 235 additional intensification, which are added to annual emissions along the stated occupation periods),
 236 respectively (see Table A1). Emission credits from post-production substitution effects (e.g. distiller's dry
 237 grains with solubles (DDGS) from corn ethanol as a substitute of soymeal), can still be accounted for in a
 238 separate step to increase transparency. Therefore, reported emissions in LUC_{global} do not include such case-
 239 specific secondary effects.

240 **2.3.2 Dynamic Baseline Methods**

241 Dynamic baseline methods assume that deforestation LUC as a consequence of the assessed product would
 242 otherwise have occurred anyway after one year, if the region has historical deforestation trends (Kløverpris
 243 and Mueller, 2012; Schmidt et al., 2015). This means that the studied crop is only ascribed the LUC
 244 emissions from deforestation corresponding to advancing them by one year. To calculate the reported

emissions with dynamic baseline methods, we apply a time-discounted version of the Global Warming Potential (GWP) factor to consider the time effect of anticipating LUC emissions by one year (Kløverpris and Mueller, 2012; Schmidt et al., 2015), which we denote as GWP_{LUC} . This is calculated as the difference between the cumulated radiative forcing (RF) of a CO_2 pulse emission over 100 years and a CO_2 pulse emission over 99 years, divided by the cumulated RF of a reference CO_2 pulse emission over 100 years (Kløverpris and Mueller, 2012) (Figure 1). Because the emitted GHG is in both of the compared cases CO_2 , the GWP_{LUC} factor can be expressed with the cumulated residence in the atmosphere only and the radiative efficiency cancels out (see Appendix B for a detailed description):

$$GWP_{LUC} = \frac{\int_0^{100} R_{CO_2}(t)dt - \int_0^{99} R_{CO_2}(t)dt}{\int_0^{100} R_{CO_2}(t)dt} \quad (1)$$

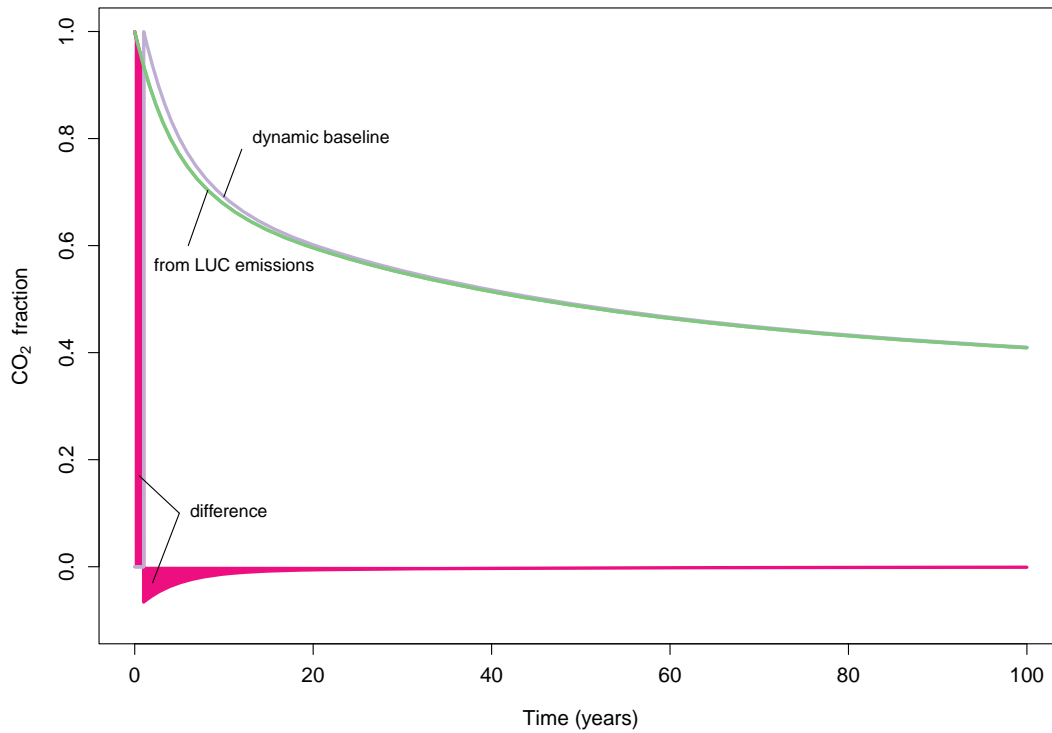


Figure 1 LUC emissions under dynamic baseline conditions. The CO_2 fraction of a generic LUC emission (green) and of the dynamic baseline (violet) over 100 years. In magenta the difference between them, 0.8%, the value of the GWP_{LUC} factor.

As earlier mentioned, two variants of the dynamic baseline method (DBM) are here considered. For the first variant (LUC_{DBM1}), GWP_{LUC} is applied to the *ad hoc* LUC emission estimates of each case study, resulting in a one-time discounting of the respective CO_2 emissions (Kløverpris and Mueller, 2012). For the second variant (LUC_{DBM2}), GWP_{LUC} is applied only to the CO_2 emissions from the expansion share of the LUC_{global} factor (i.e. $141.6 \text{ Mg } CO_2 \text{ ha}_{dem}^{-1}$; see Table A1), which results in $1.1 \text{ Mg } CO_2 \text{ ha}_{dem}^{-1}$. However, the LUC_{DBM2} does not consider land expansion emissions as a single CO_2 release that happens at the beginning of the project with the establishment of the energy crop, but rather as part of an existing annual trend (Schmidt et al., 2015). Consequently, the considered LUC emissions in LUC_{DBM2} are the sum of two global average, annual GHG emission factors that correspond to land expansion ($1.1 \text{ Mg } CO_2 \text{ ha}_{dem}^{-1} \text{ yr}^{-1}$) and intensification ($2.1 \text{ Mg } CO_{2eq} \text{ ha}_{dem}^{-1} \text{ yr}^{-1}$) effects. This results in a total of $3.2 \text{ Mg } CO_{2eq} \text{ ha}_{dem}^{-1} \text{ yr}^{-1}$ that are added to other GHG emissions in the biofuel production during the different occupation periods.

2.3.3 The GHG Protocol method

The LUC accounting method of the GHG protocol (LUC_{GHGP}) is the one explained in Annex B.2 of the product life-cycle accounting standard (Greenhouse Gas Protocol, 2011), used to estimate average LUC in products of unknown land-use and origin. In a nutshell, LUC_{GHGP} is a country-specific version of the LUC_{global} top-down model excluding intensification emissions. That is, it represents the country-average LUC emissions of demanding one additional hectare of the assessed crop (known or assumed to originate from that country). Consequently, this method is only valid to estimate LUC emissions of crops that originate (knowingly or presumably) from countries where deforestation is ongoing (i.e. Malaysia and Brazil). Indirect LUC emissions that may arise from energy cropping in US (corn) and Denmark (willow) were therefore not accounted with this method. Land-use area cover statistics from FAO stat database (2003 to 2014) are taken, considering only the new areas converted to the respective plantations (oil-palm or sugarcane). This is the consequential variant of the method, different from a possible attributional variant which would take the whole existing plantation area in the country to compute an average LUC emission factor instead (Milà i Canals et al., 2012). See Appendix C for additional details.

2.3.4 LUC emission factors from the amended RED directive

In the last RED average LUC emission factors per crop group (e.g. sugar crops, oil crops, etc.) are reported in g CO₂eq MJ⁻¹ (Appendix V, European Commission 2015). For a better comparison, the crop-specific LUC emission results were taken from the original report (Chapter 4, Valin et al. 2015), with non-amortised results given in tonnes CO₂eq ha_{dem}⁻¹. Short rotation woody crops are excluded in the amended RED of 2015 though. Being their current demand and supply marginal, related global LUC from scaling up their demand was not simulated with the agro-economic model implemented (GLOBIOM). Hence LUC emissions could not be calculated with this method for the willow case study. See calculation details in Appendix D. In Table 3 a summary of the five LUC methods is presented with their main features.

Table 3. Summary of main characteristics of the LUC methods applied in this study.

LUC method	Model type	Expansion	Intensification	Amortization
LUC _{global}	Top-down. Global average	Yes	Yes	Yes [□]
LUC _{DBM1}	<i>Ad hoc</i> LUC × GWP _{LUC} factor	Yes	No	No
LUC _{DBM2}	LUC _{global} × GWP _{LUC} factor	Yes	Yes	No
LUC _{GHGP}	Top-down. Regional average	Yes	No	Yes [□]
LUC _{RED15}	Bottom-up. GLOBIOM model	Yes	Yes*	Yes [□]

* Yield improvement, substitution and reduced demand effects considered for the calculation of final expansion, but intensification emissions not included. [□] Amortization according to the criterion presented in Section 2.1 (see Table 1).

3. Results

Taking the *ad-hoc* LUC emissions as reference, it is seen that the top-down approach LUC_{global} gives the closest LUC estimates (underlined in Table 4), followed by the LUC_{GHGP}. The latter, despite having a higher spatial resolution than LUC_{global}, showed the highest emission estimates of all methods. This may be explained by the exclusion of intensification effects in its calculation. Remarkably, the country-average LUC_{GHGP} emissions estimate for the oil-palm case is similar to the representative case of secondary rainforest clearing emissions, both LUC estimates having excluded peat oxidation and intensification emissions. LUC_{global} estimates were 28% lower, 27% lower and 34% higher than the willow, sugarcane and corn references, respectively. If we exclude the (otherwise intrinsic) intensification emissions from the LUC_{global} factor (i.e. only considering 165.5 Mg CO₂eq ha_{dem}⁻¹, see section 2.3.1) to harmonise it with the *ad-hoc* LUC for the US corn ethanol case, the two estimates differ by only 2%.

306 The accounted LUC_{RED15} emissions were systematically lower than the reference *ad hoc* LUC estimates and
307 the top-down emission factors. The reasons behind this are the included short-term effects (e.g. 42%
308 reduction of palm-oil demand), the high C-sequestration assumptions in soil and biomass (e.g. as much as
309 62% of the total CO₂eq emissions from LUC for sugarcane) and the embedded substitution effects in corn
310 ethanol (26% of new land demand covered by DDGS substitution, while 18% of feed demand is reduced), as
311 stated in Valin et al. 2015.

312 On the other hand, the LUC_{DBM1} applies a one-time, absolute discounting of 99.2% (the GWP_{LUC}) to the *ad*
313 *hoc* LUC emissions, which explains its great deviation respect to any other method. The LUC_{DBM2} applies the
314 same discounting to the share of (annual) agricultural expansion in the LUC_{global} factor, which summed over
315 the predicted occupation period results in moderate estimates (yet significantly lower than the *ad hoc*
316 references and the other top-down factors). Even though each DBM accounts for LUC emissions in a
317 different manner, they apply the same discounting logic. As a result, LUC emissions from agricultural
318 expansion in both DBM were consistently and significantly lower than any other method. The logic of DBM
319 methods and its validity are discussed in section 4.2.

320 **Table 4.** GHG emission accounting of LUC with different methods for the four biofuel study cases. Underlined the
321 LUC estimates closest to the *ad hoc* LUC estimates. DK stands for Denmark, BR for Brazil, MY for Malaysia, NA for
322 not accounted. See Sections 2.2 and 2.3 for calculation methods.

Accounted LUC GHG emissions (Mg CO ₂ eq ha _{dem} ⁻¹)						
Energy crops	<i>Ad hoc</i> LUC	LUC _{DBM1}	LUC _{DBM2}	LUC _{global}	LUC _{GHGP}	LUC _{RED15}
Willow (DK cropland)	283	2	63	<u>206</u>	NA	NA
Sugarcane (BR cropland)	311	4	95	<u>226</u>	428	43
Oil-palm (MY forest)	388	5	79	216	<u>428</u>	211
Corn (US cropland)	169	1	95	<u>226</u>	NA	12 [†]

323 [†] It represents corn from EU.

324 4. Discussion

325 4.1 Time horizons in LCA of biofuels

326 Different time horizons coexist in LCA of biofuels. The main problem related to them may be their lack of
327 definition in the LCA standards (BSI, 2011; ISO, 2013, 2006a, 2006b). As a result, most practitioners are
328 unaware of i) their differences, ii) the implicit assumptions done when carrying out a LCA, and iii) the mixed
329 effects they may have on results. For example, in an LCA of a willow plantation, these time horizons can
330 potentially overlap each other. The technological time scope would be here the lifetime of a small-scale
331 cogeneration plant (Energinet, 2012), if it is used for CHP, coinciding with the expected plantation lifetime
332 of 20 years (Saez de Bikuña et al., 2016). This time horizon would again coincide with the 20 year
333 amortization for LUC prescribed in the IPCC guidelines (IPCC, 2006) and most LCA regulations (BSI,
334 2011; European Commission, 2009; ISO, 2013; The Greenhouse Gas Protocol, 2006). If the plantation is
335 established on a normal arable or grassland where no long-term emissions are expected beyond the 20 year
336 rotation period, the inventory modeling period would be the same as the previous time horizons (20 years).
337 Finally, a 20-year impact modeling period could be picked to calculate the relevant GW impacts (i.e.
338 GWP₂₀), when really short-term effects are to be represented (in this case over the next two decades only;
339 UNEP/SETAC Life Cycle Initiative, 2016). But these time horizons can also be different from each other
340 (and most often are). For example, a ‘farm-to-gate’ LCA of a land-based product may stick to the crop
341 lifetime (e.g. six year for a sugarcane plantation) to define the expected occupation period (Seabra et al.,
342 2011), since there is no power-plant or biorefinery inside the system boundaries. If the crop is established on
343 a drained peatland, there will be GHG emissions beyond the crop’s lifetime which need to be accounted.
344 This would result in four different time horizons: a 6 year occupation period, an arbitrary 20 year
345 amortization period (according to most regulations), an extended inventory modeling period (e.g. 50 years,
346 Valin et al., 2015) to account for the long-term peat oxidation and an arbitrary impact modeling period to
347 calculate the GW impact (e.g. 100 years, i.e. GWP₁₀₀).

348 These two examples illustrate the conundrum around time horizons in LCA of biofuels. If these time
349 horizons are properly distinguished, extending the inventory modeling period to account for long-term

emissions (e.g. peat oxidation) in biofuel LCAs would not imply extending its amortization period (Valin et al., 2015). This is because, as it has been argued, the technical basis for amortization is the lifetime of the involved technology, which is independent from any author's subjective standpoint. Therefore, according to our definition proposals, including long-term emissions in LCA of biofuels established on peat land would increase the annualized LUC emissions (and related GW impacts), not decrease them (Valin et al., 2015). Likewise, changes in SOC and foregone C-sequestration (which are caused by cultivation) need to be considered for the identified inventory modeling period (which is the expected occupation period given by the amortization criterion established in Section 2.1), not less (U.S. EPA, 2010). In this respect, the 30 years amortization choice stands not as an arbitrary one (Plevin et al., 2015), but as an engineering and financial criterion generally applied in industry. On the other hand, the full-rotation period of tree cultivars could be a better estimate of the transformed land's long-term occupation in forestry systems (as it would represent the minimum commitment period of the responsible company). But forestry systems are multi-output systems which companies optimize according to market price of different products, 'bioenergy' being more often a by-product (Cintas et al., 2015), unlike the agricultural study cases analyzed herein.

It is thus evident that defining and distinguishing the time horizons mentioned in Section 2.1 is a prerequisite to perform biofuel LCAs in a more transparent and consistent way. Nevertheless, the choice of the applied modeling periods for the characterization of (e.g. GW) impacts may be done by convention (e.g. GWP₁₀₀), but remains arbitrary as it implies value judgements over the importance given to short-term and long-term (GHG) emissions (Brandão and Canals, 2012). GHG inventories can be though discretized in annual steps and be combined with dynamic characterization factors to calculate time-adjusted GW impacts (Levasseur et al., 2010; O'Hare et al., 2009), which would be the most accurate solution to deal with LUC and other life-cycle GHG emissions of land-demanding products. These methods were developed to avoid the bias of traditional LCA, where all life-cycle emissions are summed up and the GWP is calculated from the aggregated score (i.e. all emissions are assumed to occur in the first year). These methods were not applied in this study because it focused exclusively on LUC and not the total life-cycle emissions.

375 4.1.1 Long-term emissions and post-occupation LUC in biofuel assessments

376 Post-occupation LUC may be also considered (also referred to as post-cultivation LUC, Sanchez et al. 2012).
377 These can be confused with ‘pure’ long-term emissions though. While ‘pure’ long-term emissions from the
378 slow decay of previously accumulated organic matter (e.g. peat oxidation) start with the establishment of the
379 biofuel project (and may continue for decades Fargione et al. 2008; Valin et al. 2015), post-occupation LUC
380 emissions start at the end of the biofuel project, when the biorefinery or power-plant is no longer in
381 operation. These will be thus the result of future human land-use activities on the released land (e.g. natural
382 regeneration, if the land is abandoned). This differentiation is important because physical or ‘pure’ long-term
383 emissions in dedicated biofuel LCAs regard almost exclusively energy crop plantation establishments on
384 drained peat land, which mainly take place in Indonesia and Malaysia (Valin et al., 2015). On the contrary,
385 post-occupation LUC emissions may apply to any biofuel (and land-demanding product), while being
386 completely determined by the global (or regional) land market and land-use trends at the end of the identified
387 amortization (i.e. occupation) period. Since “land use after 30 years is highly uncertain and there is no
388 guarantee of future rotations” (U.S. EPA, 2010), the exclusion of post-occupation LUC (which are emissions
389 from future human land-uses) in dedicated biofuel LCA seems desirable. When post-occupation LUC
390 assumptions are unavoidable in biofuel LCA (e.g. regarding the fate of oil-palm trees at the end of the
391 occupation), practitioners should depict key choices as different scenarios and/or test them in a sensitivity
392 analysis. Most energy crops are nonetheless entirely harvested to obtain the final biofuel product. This is the
393 case for all the starch-rich and sugar crops used to produce ethanol, annual oil crops to produce biodiesel
394 (Reijnders and Huijbregts, 2008) and short-rotation perennials like willow, poplar, switchgrass or miscanthus
395 to produce biomass for energy purposes through co-digestion (Tonini et al., 2012), co-incineration (Heller et
396 al., 2004) or gasification (Saez de Bikuña et al., 2016). That is, the vast majority of energy crops are fully
397 removed at the end of their rotation periods. If the assessed energy crops have short-rotation periods (and
398 most of them do, like corn, sugarcane or willow), the grounds to credit any temporary C-sequestration
399 potential beyond SOC gains is questionable (Cherubini et al., 2011). In fact, assuming the frequently
400 harvested biomass as a permanent C sink during the occupation period may underestimate the real GW
401 impact potential (U.S. EPA, 2010). In their analysis (Figure 2.4-36, page 392), EPA assumes one cut every 3

402 years for sugarcane with an average (but constant) C-sequestration equivalent to one-year growth. However,
403 this may contrast sharply with the Brazilian reality of 5 cuts in a 6-year cycle as described in Macedo et al.
404 2008. Such long-term emission assumptions may dramatically affect reported benefits, resulting in high C-
405 sequestration estimations when combined with high productivity crops like sugarcane despite questionably
406 representing reality.

407 Therefore, if long-term emissions (e.g. from peat oxidation) are not expected, the inventory modeling period
408 in LCA of land-demanding products like biofuels should be restricted to the expected occupation period, so
409 as to be in line with the accuracy, completeness and relevance accounting principles (ISO, 2006b).

410 **4.2 Dynamic baseline methods for GHG emission accounting from LUC**

411 Proponents of DBM do not specify the time horizons over which their discounted iLUC factors may be
412 considered. In an endeavour of obtaining ‘amortization-free’ iLUC emission factors, it is ignored that the
413 occupation time definition is unavoidable in LCA (Martin, 2013). But assuming a dynamic land-use baseline
414 brings more problems. First and foremost, assuming that the global food supply and demand are not reduced
415 in the long-term, the (additional) occupation of arable land is the primary cause that triggers iLUC effects
416 (Kløverpris, 2008; Schmidt et al., 2015). This means that, as long as there is land occupation for energy
417 cropping with food production displaced elsewhere, a cause-effect link (the iLUC causality, see Appendix
418 E.1) is established between the two activities. In other words, the production of this additional food to meet
419 the new demand is linked to the initial land occupation (in terms of carbon and nitrogen flows, as far as GHG
420 emissions are concerned). That is, this iLUC causality link must last as long as land occupation for energy
421 cropping lasts (see Appendix E).

422 DBMs also discount LUC emissions, which do not need to be reported by those who convert natural land and
423 these will thus remain unaccounted for (see Figures E1 and E2). Not accounting for such emissions implies
424 that the responsibility of the impacts derived cannot be ascribed to anyone and it hence violates the LCA
425 founding polluter-pays principle, as well as the completeness, relevance, transparency and accuracy
426 principles of GHG accounting (Greenhouse Gas Protocol, 2011; ISO, 2006b).

427 Second, proponents of DBM take deforestation as a given boundary condition, rather than recognizing it as a
 428 consequence of the same or similar (anthropogenic) processes that they assess. This ignores the actual
 429 interdependence between the assessed project and the dynamics of the dynamic baseline. The
 430 interdependence between the anticipated dynamic baseline and the assessed project lies in the effect that the
 431 project will itself increase the reference LUC level for other similar future projects. This is in conflict with
 432 the requirement that reference and project must be independent to avoid logical circularity and, practically,
 433 leads to the phenomenon that the discounted amount of GHG emissions will increase with the global rate of
 434 LUC. Similar mechanisms have been observed in fishery and conservation sciences, where this phenomenon
 435 is described as shifting baseline (Papworth et al., 2009; Pauly, 1995). Making the baseline dynamic results in
 436 a positive feedback propelling LUC. This is reflected in the low fractions of real LUC GHG emissions that
 437 are actually accounted for by the DBMs (see Table 1 and Table E1 and Figure E1 in Appendix E).

438 Third, rather than a single dynamic land-use baseline (e.g. a dynamic natural regeneration baseline, Milà i
 439 Canals et al. 2007; Soimakallio et al. 2015), DBM suggest a double land-use baseline of two different land-
 440 use systems (i.e. a natural forest cover in steady-state and an unknown human land-use system) that overlap
 441 in time. Consequently, the dynamic land-use baseline cannot be a ‘business-as-usual’ (BAU) baseline that
 442 would apply in a consequential LCA (Soimakallio et al., 2015). BAU are single land-use baselines which are
 443 either known (previous land-use) or estimated (marginal land-use) and they are included from year zero (not
 444 at year ‘one’) in the system boundaries of the LCA.

445 The following analogy illustrates the effect and makes it subject to logical reasoning. Taking regional
 446 deforestation trends as dynamic land-use baselines is similar to applying the current GHG emission pattern
 447 as dynamic atmospheric baseline to account for the global warming (GW) impacts of additional fossil fuel
 448 combustion (Saez de Bikuña et al., 2016). This would mean accepting climate change to happen as a baseline
 449 instead of pre-industrial climate conditions that are usually regarded a natural reference for climate change
 450 (Hartmann et al., 2013) and the only proven safe climate space for human development (Steffen et al., 2015).
 451 Following the dynamic baseline logic, the GHG emissions of burning 1 Mg of *additional* oil could then be

452 considered as “anticipated fossil oil combustion” and accounted as 0.3 Mg CO₂e (with a GWP of 0.8%, see
453 Figure 1) instead of 34.9 Mg CO₂eq (see Appendix F.3).

454 The above arguments have led us to reject the current DBMs. This brings, however, the amortization
455 problem –and the related the long-term occupation period– into focus again. Amortization of upfront
456 emissions (e.g. from a power plant installation) has not been problematic before, because these tended to be
457 negligible from a life-cycle perspective. LUC are, however, a special type of upfront emissions that have
458 become controversial precisely because of their significant magnitude. It has been shown that early GHG
459 emissions have a more important role in the GW impact of biofuels than later emissions or removals (O’Hare
460 et al., 2009). The amortization of LUC emissions can only be avoided by calculating a time-adjusted GHG
461 emission inventory combined with time-dependent GW characterization factors (Levasseur et al., 2010), but
462 this requires determining the occupation period over which the assessed land will be used. In this respect, the
463 proposed technical criterion to determine the occupation period is dependent on the biorefinery or biofuel
464 type, rather than the practitioner. Therefore, it can minimize the critical role subjects play in determining the
465 environmental impacts of biofuels (Kløverpris and Mueller, 2012; Sanchez et al., 2012).

466 **4.3 Implications for future LUC modeling and policy-making**

467 LUC can be seen as the environmental investment that biorefineries and power plants using the feedstock
468 need to pay off through their biofuel production. Ascribing LUC to biorefineries and power-plants could
469 facilitate the tracking of LUC emissions, as it renders the pollution responsibility clearer. This may open the
470 opportunity to legally bind LUC emission reporting to biorefineries and power-plants, which would need to
471 report on a regular (e.g. annual) basis the remaining C debt (Fargione et al., 2008) they are obliged to pay
472 back. This way, the real C debt could be calculated and be paid off with the exact amount of biofuel
473 produced/sold. To ensure the claimed benefits of biofuels (from fossil fuel substitution), the total energy
474 demand of the relevant sectors in the country it is consumed must be constant to avoid leakage or rebound
475 effects (Druckman et al., 2011; Hertwich, 2005; Lambin and Meyfroidt, 2011), while cross-checking the C-
476 debt with an independent accountant or third party.

477 Importing energy crops or biofuels from (economically) abandoned land origin poses serious certification
478 challenges. Economically abandoned and marginal lands are contingent and may change with changing
479 social dynamics (subsidies, new cultivars and technology, etc.) (Hatna and Bakker, 2011). However,
480 importing biofuels from physically degraded land (e.g. former tropical forestland in Indonesia that has been
481 invaded by alang-alang grasses) seems more plausible (Searchinger and Ralph, 2015), since those grasslands
482 can be tracked and monitored with satellite imaging systems. For abandoned farmland, it might be more
483 appropriate to incentivize their use whenever a more direct control and monitoring can be established
484 (Pointereau et al., 2008; Terres et al., 2013) and which could be combined with existing relevant policies
485 ((EC), 2003; EEA (European Environment Agency), 2009) to tackle rural depopulation and biodiversity loss.

486 Even though yield estimates are still needed to compare the environmental performance of biofuels to fossil
487 fuel counterparts in LCA, the main uncertainty source of LUC emission factors in economic models (the
488 yield related parameters) (Plevin et al., 2015) could be avoided. If an accounting or certification system is
489 established that ascribes LUC emissions to companies responsible of the land clearing (measured on site, per
490 ha expanded, or estimated through top-down models), calculated and reported on an area basis instead of an
491 energy (or product) basis (see Table 4). This accounting system would be more in line with a corporate
492 reporting system than a project or product accounting system (Greenhouse Gas Protocol, 2004), and the
493 companies holding the C debt from LUC would pay it back retrospectively, i.e. with the produced biofuel.
494 These figures would be estimated on the basis of factual, recorded yield data rather than on predicted and
495 uncertain yield estimations.

496 Moreover, reporting LUC emissions per ha can also avoid possible confusions behind the biofuel emissions'
497 calculations if reported on an energy basis (e.g. LUC reported per energy content –gross energy output – or
498 per useful energy –net energy output), when different technologies are at hand to provide the same service
499 (e.g. heat and power). Under this new perspective, the environmental performance of different land use
500 systems which produce biofuels is assessed (as in e.g. Tonini et al. 2012), rather than biofuel products in
501 itself.

502 The lack of transparency behind economic LUC models contributes to their uncertainty, represented by the
503 broad range of results in literature. For instance, it is not clear whether economic iLUC models, despite
504 including intensification effects, always include the related GHG emissions that would derive (directly and
505 indirectly) from an increased used of fertilizers to meet the simulated demand shock (U.S. EPA, 2010; Valin
506 et al., 2015).

507 The simplicity of top-down models gives them a special advantage over the sophisticated and complex
508 bottom-up models: higher transparency. These can be easily examined, reproduced and updated, contrary to,
509 e.g., the ‘black-box’ economic LUC models (Broch et al., 2013). Top-down LUC emission factors represent
510 mean GHG emissions from demanding additional land globally (LUC_{global}) or regionally (LUC_{GHGP}),
511 depicting average agricultural expansion and intensification emissions that new land-demanding products
512 like the studied biofuels generate. Their simple calculation does not impede the inclusion of possible
513 substitution effects (like, e.g., DDGS from corn ethanol) in another step of the assessment. Results could be
514 easily revised periodically with updated land-use cover and synthetic-N production statistics, while their
515 uncertainties would mainly relate to C stock estimates of the affected land areas. Despite being less
516 representative of the product-specific LUC effect than bottom-up models (e.g. LUC_{global} does not
517 differentiate between demanding corn or wheat, in China or in US), top-down models can provide an
518 external reference system for a rough validation of bottom-up LUC estimates (e.g. they can provide a scale
519 reference to determine the right order of magnitude of LUC emission estimates). Such top-down LUC
520 models can theoretically converge to (and in the limit coincide with) national and global LUC emission
521 statistics and thus be a way to reconcile bottom-up assessments (Creutzig et al., 2012), providing
522 complementary information for regulation and policy-making (European Commission, 2015). The authors
523 recognize that deforestation and LUC phenomena are intricate processes which involves several actors and
524 which spans over several years (Gaveau et al., 2016). Nevertheless, top-down LUC factors can be a way to
525 set the mean of regional and global LUC emissions by taking a broad landscape approach and by simplifying
526 a complex issue that requires urgent action.

527 **5. Conclusions**

528 Different methods have been investigated to explore their advantages and disadvantages in the accounting of
529 GHG emissions from LUC in biofuel LCAs. To ensure result harmonization, the different time horizons
530 involved in LCA have been defined and the methods have been applied to four known biofuel study cases.
531 The main findings of the study can be summarized as follows:

- 532 • Six different time horizons (technological scope, inventory model, impact characterization,
533 amortization/occupation, plantation lifetime, harvesting frequency) can and should be distinguished
534 when performing an LCA of land-demanding products. Apart from the impact characterization, the
535 other time horizons can be defined according to an exclusively technical criterion. It is of crucial
536 importance to agree, first, on their definitions and secondly, on their values (for LCA studies of same
537 biofuels/products), to allow full comparability and foster high quality environmental footprinting
538 standards.
- 539 • The reasons behind systematic underestimations (dynamic land-use baselines and high C-
540 sequestration assumptions) and overestimations (exclusion of intensification effects) of LUC
541 emissions were identified. In this regard, the validity of the large discounting (99%) applied by
542 current DBM to account for deforestation emissions is disputed. Even though the amortization of
543 LUC emissions can be avoided with time-adjusted GHG inventory and GWP characterization
544 factors, defining the inventory modeling period (i.e. land occupation period) cannot. This is an
545 intrinsic part of the system boundaries of any LCA dealing with land-demanding products and the
546 cut-off criterion should be clearly stated in the scope phase.
- 547 • The technical lifetime of biorefineries or power-plants (technological scope) in LCA of land-
548 demanding biofuels is proposed as the best proxy for this cut-off criterion, which it is claimed to
549 represent the long-term occupation of the land used for the production of feedstock. This is a purely
550 technical –thus robust– criterion, valid to determine both the amortization (if performed) and
551 inventory modeling periods. This is in line with common economic practice in industry for the

amortization of investments and the calculation of the net present value, thus used for project management and decision-making.

- Calculating LUC emissions per ha and legally binding them to companies responsible for the agricultural land expansion could avoid the uncertainty related to yield estimation in LUC emission estimates.
- Top-down models represent average LUC emissions at different spatial resolutions derived from deforestation statistics. Their results can serve as a rough validation reference for bottom-up LUC emission estimates, being complementary rather than competitors for decision-support and policy-making.

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