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Indirect Land Use Change from biofuel production: implications for biodiversity

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The UNEP World Conservation Monitoring Centre (UNEP-WCMC) was established in 2000 as the world biodiversity information and assessment centre of the United Nations Environment Programme. The roots of the organisation go back to 1979, when it was founded as the IUCN Conservation Monitoring Centre. In 1988 the World Conservation Monitoring Centre was created jointly by IUCN, WWF-International and UNEP. The financial support and guidance of these organisations in the Centre's formative years is gratefully acknowledged.

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Contents

1 Biofuel development and iLUC

Biofuels (i.e. fuels produced from biomass) and their use as a source of renewable energy have developed rapidly since the beginning of the 21st century. Global consumption of first generation biofuels almost tripled from 28 billion litres in 2004 (IEA 2006) to 81 billion litres in 2008 (66 billion litres of ethanol and 15 billion litres of biodiesel, Fargione *et al* 2010). This trend is likely to increase.

Countries aiming at increasing the share of energy consumption from renewable sources have set targets for the share of transport fuel to be supplied by biofuels in the future. For instance, the European Union agreed to a mandatory 10% minimum target to be achieved by all Member States for the share of biofuels in transport petrol and diesel consumption by 2020 (Commission of the European Communities 2009). In order for countries to achieve such targets, biofuel production needs to further increase.

The growing demand for biofuels means an increase in the land required for their cultivation. It is estimated that the land used for bioenergy production increased between 2004 and 2008 from 13.8 million hectares (Mha) (IEA 2006) to 33 Mha, and in 2008 represented about 2.2% of global cropland (Fargione *et al* 2010). Meeting future demand will require further expansion, and the question of where this expansion is likely to take place is the subject of some debate. While the current main producers of biofuels, the USA, EU and Brazil (Table 1), can be expected to remain key players, plans for the development of bioenergy especially in China and India suggest that these countries will play a larger role in the future (Gallagher 2008), and changes in the types of biofuels used mean that other countries such as Indonesia and Malaysia, by far the largest producers of palm oil, may also increase their roles.

Table 1. Main biofuel producing countries, main crops and land requirement in 2008 (adapted from Fargione *et al* 2010).

Country	Main bioenergy crops	Land required in 2008 (Mha)
United States of America	Corn and soy bean	13.07
European Union	Oilseed rape/ canola	9.40
Brazil	Sugarcane	6.04
Argentina	Soybean 1.79	
China	Corn	1.04
Canada	Corn and wheat	0.49
Indonesia/Malaysia	Palm	0.33

There are two principal avenues for increasing the land available for producing biofuel feedstocks without adverse impacts on other forms of agricultural production:

a Through conversion to cropland of land not currently under agricultural production (including pasture).

b Through intensification of production on existing agricultural land so that biofuel crops can be grown while food crop yields remain the same and no further land is converted.

Both of these forms of land use change can occur either as a direct result of biofuel production, occurring where the biofuel crop is cultivated (direct land use change), or indirectly, due to some other form of production displaced by biofuels. In the latter case, the phenomenon is called **indirect Land Use Change**, or **iLUC,** and concerns have been raised that it may increase the overall impacts of biofuel development and that associated greenhouse gas (GHG) emissions may reduce considerably the effectiveness of biofuel use in reducing overall GHG emissions (see also, e.g. Croezen *et al* 2010; Edwards *et al* 2010; Fritsche & Wiegmann 2011; Searchinger *et al* 2008). Indirect land use change can take place through two principal mechanisms:

1.1 "Conversion iLUC"

Conversion iLUC occurs when a biofuel crop replaces a food crop (or the use of a crop switched from food to fuel) and production of the food crop is displaced to another area of land, where it may in turn displace some other land use. This means that potentially a whole chain of displacements can be triggered, eventually causing conversion to agriculture of an area that was not previously under agricultural production (Figure 1). The distance between the location of a change from a food to a fuel crop and the location of conversion of non agricultural land to agriculture can be very large. Therefore, identifying and documenting conversion iLUC can be very difficult.

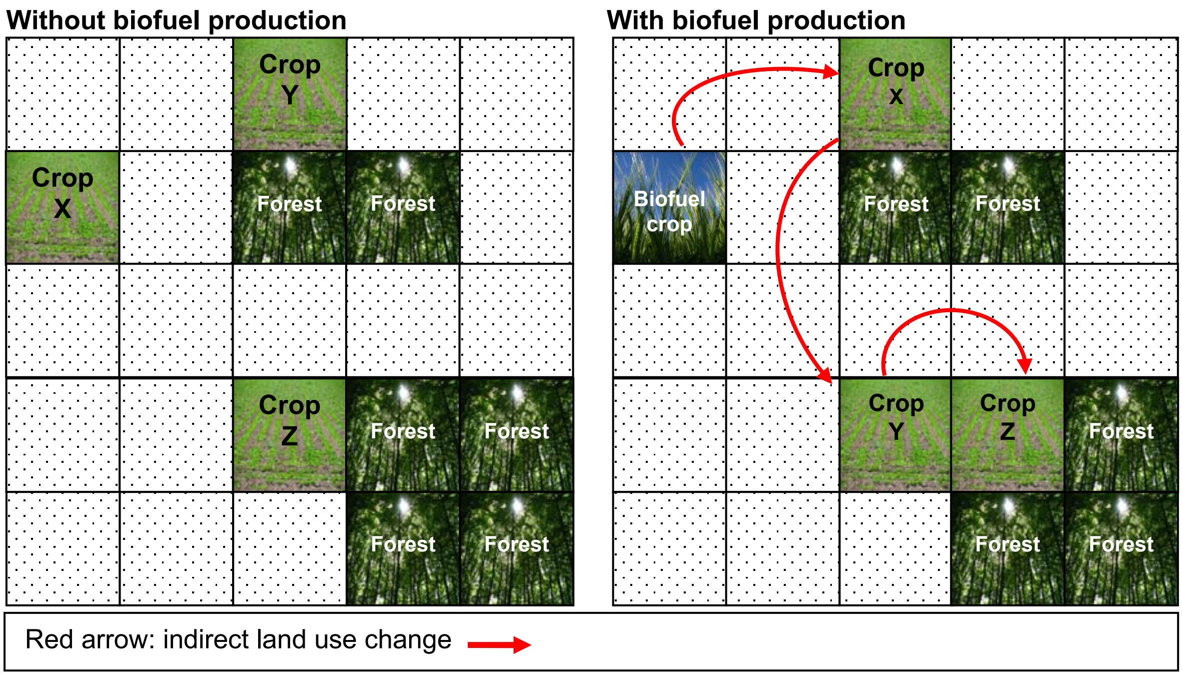

Figure 1. Schematic representation of "conversion iLUC" in a landscape matrix; the conversion of a unit of agricultural land from food to fuel production displaces the initial land use (and potentially others) may ultimately lead to conversion to agriculture of previously non-agricultural land.

1.2 "Intensification iLUC"

Intensification iLUC occurs when conversion of an existing agricultural area to biofuel crop cultivation causes agriculture in remaining production areas to be intensified in order to maintain overall food crop yield without expanding the total area cultivated (Figure 2). The place where intensification occurs can be far away from the place where conversion to biofuel production has occurred.

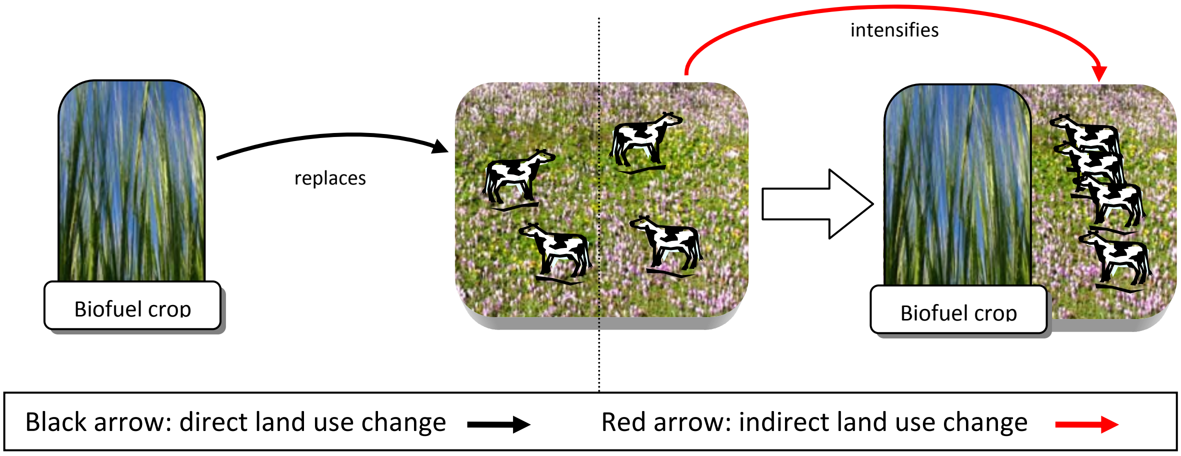

Figure 2. Schematic representation of "intensification iLUC"; the conversion of a unit of agricultural land from food to fuel production without expansion of the total production area means that production is intensified in the remaining food producing area.

Conversion iLUC is widely recognised and discussed, especially in the context of the additional greenhouse gas (GHG) emissions it can cause (e.g. Fritsche *et al* 2010; Overmars *et al* 2011; Plevin *et al* 2010) and their implications for the achievement of emissions reduction targets. However, intensification iLUC has received far less attention. In fact, intensification has even been suggested as an important solution to iLUC rather than a separate form of it (Stehfest *et al* 2010). While intensification can reduce overall land required for agriculture and pasture, thus potentially avoiding some conversion of land not currently used for these purposes, the increased agricultural inputs it requires have major GHG implications, and intensive production methods can have significant impacts on biodiversity (see below).

In practice, iLUC of either type is very difficult to identify and document. The factors that contribute to this complexity include:

- Land for cultivation of biofuel crops can be gained through combinations of direct and indirect land use change, including both conversion and intensification;
- In both types of iLUC, the location where iLUC occurs can be a long way from where the new biofuel cultivation takes place. Original land uses can even be displaced to another country or another continent;
- In conversion iLUC, area of land involved can vary along the ‘conversion chain’, depending on the fertility of land and agricultural practice in different locations and on requirements of different food crops. This can cause the ‘conversion chain’ effectively to split into several that are even more difficult to trace;
- The global area under agricultural production is increasing continually due to increasing demand for food as well as to the growing demand for biofuels. It can be very difficult to ascertain whether an area has been converted as a result of biofuel feedstock cultivation or because of the expansion of agricultural production area for food crops.

These complexities have made it impossible, to date, to arrive at a full picture of where iLUC has happened already and how much area has been affected, where it will happen in the future, and what its implications for biodiversity are. Some of the existing information on iLUC is summarised in the following.

2 Evidence for and projections of iLUC

Due to the above complexities of iLUC, concrete evidence showing where iLUC has actually occurred is very scarce. Web searches using Google and Google Scholar found neither scientific nor grey literature on evidence for iLUC. The only evidence for specific instances of iLUC can be derived by drawing on several different sources for empirical data on cultivation area and production. Box 1 provides examples for how production statistics and other information can be used to deduce iLUC.

As iLUC is essentially invisible, i.e. it will hardly ever be possible to attribute the conversion of a piece of land in one place to the use of crops for biofuels from another piece of land, iLUC is mainly assessed with the use of models. There is as yet no empirical evidence of land use change, as it would depend on detailed tracing of money, land use and agricultural products, but statistical data on recent changes in production have been used to infer iLUC in at least two cases (see Box). The remaining evidence for iLUC is derived from models. Using models – mathematical representations of complex systems – the iLUC of biofuel production can be calculated as the difference in the area covered by land uses other than biofuel crops between scenarios with and without biofuel production (whereas direct LUC is the change in the area covered by biofuel crops under the two scenarios). Various models have been used to estimate the iLUC impacts of biofuel production, mostly in terms of GHG emissions (Searchinger *et al* 2008), but only very few models exist that map iLUC spatially explicitly (Lapola *et al* 2010).

Box 1. Examples for the use of production statistics and other information sources to deduce the potential for iLUC**.**

Brazil

In Brazil, the expansion in sugarcane production areas in recent years mainly happened in Saõ Paulo state (Sparovek *et al* 2007). Of the land that was converted to sugarcane in 2007 and 2008 in this state, 45% was previously used as rangeland (Zuurbier & van de Vooren 2008). In the Amazon region, more than 90% of the soybean plantations planted after the implementation of the 2006 moratorium replaced rangeland (Lapola *et al* 2010). In addition to the increase in biofuel production area, cattle ranching is also increasing in Brazil and the expansion of rangeland is considered an important driver of deforestation (Lapola *et al* 2010). This information jointly suggests that the expansion of areas for biofuel production contributes to indirect land use change in other parts of Brazil.

Ukraine

The Ukraine has very high potential for bioenergy production (Londo *et al* 2007). It is suggested that Ukrainian rapeseed will represent the most likely source of future imports into the EU (Bauen *et al* 2010). Between 2004 and 2007 the area under rapeseed production in the Ukraine increased fivefold and then doubled again between 2007 and 2008 (Borysov 2011). In 2008, an estimated 1.4 million hectares of land were covered with oilseed crops (Bauen *et al* 2010). This development has been greatly supported by the increasing demand for biodiesel in the EU as well as national government programmes linked to subsidies (Elbersen *et al* 2009). The general assumption is that the expansion of agriculture in the Ukraine will target the large areas of land that were abandoned when the Soviet Union collapsed (Lambin & Meyfroidt 2011), thereby avoiding conversion of land not yet under production, such as forest. Moreover, Ukraine's agriculture is increasingly modernised and intensified (Kyryzyuk 2010). Intensification iLUC as well as a shift from food to fuel crops in some places is likely to occur as part of this development. Moreover, where the expansion targets areas that were abandoned for about 20 years, their conversion back to agricultural land will have an impact on biodiversity.

Meaningful comparisons among model outputs are difficult, because each model uses different policy scenarios, mixes of biodiesel and ethanol, and models different components, has a different time horizon and produces different outputs (e.g. total area vs. area per unit energy). To overcome some of these challenges, the European Commission Joint Research Centre conducted two comparisons of models to assess iLUC GHG emissions (Burrell 2010, Edwards *et al* 2010). The comparison by Edwards *et al* (2010) included six partial and full equilibrium models (AGLINK-COSIMO, CARD, IMPACT, G-TAP, LEI-TAP, and CAPRI) which were standardised as much as possible and run for four scenarios (considering EU and US biodiesel and ethanol targets). Edwards *et al* (2010) found that iLUC caused by increasing use of biofuel within the EU and US would largely occur in other regions. Depending on model and scenario, iLUC ranges from 0.1 to 1.9Mha extra cropland per extra million tonne-of-oil-equivalent (Mtoe). The large range of iLUC area projected by the various models was due to the biofuel considered (on average ethanol tended to have larger iLUC impacts than biodiesel), the geographic focus of the model (e.g. the highest iLUC was projected for biodiesel produced within Germany by the LEI-TAP model which forces all production to occur within the EU), the consideration of by-products (e.g. LEI-TAP underestimates by-products), and the yield improvements assumed to occur in non-biofuel cropland. The models compared by Edwards *et al* (2010) assume price-induced yield improvements based on extra fertilizer use that are unlikely to be realised, as marginal land brought into arable use (especially in developing countries) will likely be less suited for cultivation than existing agricultural lands. Therefore the models are likely to underestimate direct and indirect land use change (Edwards *et al* 2010, Bowyer 2010). By combining the outputs from this model comparison with National Renewable Energy Action Plans (NREAPS) of EU countries, Bowyer (2010) estimated the iLUC of biofuels and bioliquids to be between 4.1 and 6.9 million hectares globally in 2020 due to additional EU biofuel demand stimulated by the Renewable Energy Directive (RED). Most of this iLUC is anticipated to occur in countries outside the EU, because of the large share of imported biofuel or biofuel feedstock. The findings highlight the potential magnitude of future iLUC globally caused by EU policies.

Focussing on EU iLUC, Burrell (2010) compared three partial equilibrium models (AGLINK-COSIMO, ESIM and CAPRI) using two scenarios (baseline: 10% target within EU for transport energy from first and second generation biofuels, counterfactual scenario: no mandatory target), and concluded that total cropped area will be higher with biofuel policies. However, none of the models reviewed provides a comprehensive and realistic picture of indirect land use change. In particular, none of the models considered the sustainability criteria of the Renewable Energy Directive (RED; see below) and hence permitted land use change in all areas; and the assumptions about the use of second generation biofuels might be inaccurate, e.g. no land use implications and the timing of market entry might be over optimistic. Further, one of the models (AGLINK-COSIMO) assumes a fixed total agricultural area within the EU and hence not permitting expansion of agricultural area; although within the EU where agricultural areas is predicted to decrease this might not be as important. The impact of EU policies is to slow down the long-running declining trend in EU agricultural area, and biofuels or their feedstocks need to be imported. Overall Burrell (2010) concluded that "none of the models whose results are reported in the study includes all the features that could be considered desirable for the particular research question, and each model has its own particular strengths and weaknesses, the results of the three models taken together give a composite, multi-layered picture, albeit one that requires sensitive interpretation."

The above models provide regional summary statistics, but do not identify the location of land use change. Studies that provide spatially explicit information tend to focus on direct land use change. Hellmann and Verburg (in press) modelled biofuel development in Europe in response to the EU RED in a spatially explicit way as part of an assessment of impacts on land use and biodiversity. The study highlights geographical hotspots for biofuel production within Europe and makes an attempt at differentiating direct and indirect land use change. Four scenarios based on different levels of globalisation and regulation and two policy variants (without and with the RED) were applied for the period 2000-2030. The study suggests that direct effects of the RED on land use will be small, but indirect effects may be considerable. The areas that will be mostly affected by land use change are areas with semi natural vegetation, whereas forest areas are projected to increase. The area of semi natural vegetation projected to be lost to iLUC due to biofuel production is slightly higher than that projected without RED (range for four scenarios: 8,813,400 - 15,364,200ha versus 6,510,900 - 12,881,900ha). Under all four scenarios, a geographical concentration of biofuel crop cultivation is projected for parts of eastern Germany, France, Spain, Poland, Lithuania, Czech Republic, Slovakia, Austria, and Hungary. Overall this model projects substantial increases in forest area by 2030, achieved by intensification and importing crops and meat from outside the EU. Using this model to look at global impacts of the EU RED, Banse *et al* (in press) project that most biofuel crop production will expand into South and Central America.

We are aware of only one example where iLUC from biofuels has been modelled spatially. Lapola *et al* (2010) used a spatially explicit modelling framework to project the direct and indirect land use change arising from the fulfilment of Brazil's biofuel production targets for 2020 together with increases in food and livestock demand. Using the LandSHIFT model at 5 arc minute (roughly 8km) resolution, Lapola *et al* showed that biofuel plantations would largely replace rangeland areas, hence the direct land use change is small, at least in terms of carbon emissions. However, the iLUC could be as large for soybean and larger for ethanol than the direct land use changes (factor of 1.15 direct to indirect LUC in terms of area), in particular due to expansion of rangeland into forest (12,197,000ha) and other native habitats (4,600,000ha). The iLUC caused by biofuels could be avoided by increasing the livestock density from 0.09 head per ha to 0.13 head per ha. The model outputs also rely on assumptions of yield increases in soy and sugarcane.

Models predict LUC due to biofuel production within nations or larger regions, which then must then be mapped to specific ecosystem types to predict GHG or biodiversity consequences. In general, most models assume that such LUC will occur at the agricultural frontiers, indentified from historical patterns. The assumptions used to map economic model results to specific land cover classes are one of several sources of uncertainty in iLUC predictions. Most models of agricultural production assume yield increases through technological progress and intensification when calculating the needed increases in agricultural area. However, we were unable to find a study on iLUC from biofuel production that considers intensification as a separate form of iLUC. Further, most models provide output in terms of total land use change and only one modelling study (Lapola *et al* 2010) provides information on iLUC explicitly.

While the overall understanding of where exactly iLUC is going to happen is still limited, the existing modelling results suggest that iLUC is likely to grow considerably in magnitude in the future, within and outside of Europe. However, apart from these modelling results, there is one more reason for concern about the future role of iLUC. This concern arises from the fact that existing and developing sustainability standards and criteria for biofuel production are to date unable to avoid iLUC in ecosystems that are not of high carbon value – and thereby encourage it.

3 Implications for biodiversity

To date, discussions about iLUC, have mainly focused on its potential to cause additional carbon emissions (Melillo *et al* 2009; Overmars *et al* 2011; Plevin *et al* 2010; Searchinger *et al* 2008) and therefore diminish the emissions reductions that can be achieved by using biofuels. The potential impacts of iLUC on biodiversity, which are substantial, have received far less attention.

It has been widely recognised that biofuel cultivation in general can have negative impacts on biodiversity (Fargione *et al* 2009; Fargione *et al* 2010; Gallagher 2008; Phalan 2009; Stromberg *et al* 2010; van Oorschot *et al* 2011). These impacts include changes in species populations, disappearance of some species and colonisation by others, and changes in ecosystem diversity and function as well as in habitat composition and quality (Figure 3). These are consequences of changes in both land cover and land management and therefore depend both on the status of the land before it is used for biofuel cultivation and on the requirements and management of the biofuel crop (Box 2).

Box 2. Example for implications of conversion on freshwater ecosystems.

Fargione *et al* (2009) summarised the implications for freshwater ecosystems of the conversion of grassland to corn in the US. The causal chain begins with a disturbance of the capacity of intact grassland to retain soil and nitrogen. One potential consequence is sedimentation in freshwater systems, causing increased turbidity and water temperature and consequently degrading habitat for coldwater fish. Another potential consequence is that nitrate enters the freshwater system, leading to algal blooms and hypoxia. In addition, where corn requires irrigation, this is likely to have serious impacts on the water availability in and around the production area. For irrigated corn in the US, Aden (2007) calculated that 785 litres of irrigation water are needed for every litre of ethanol produced.

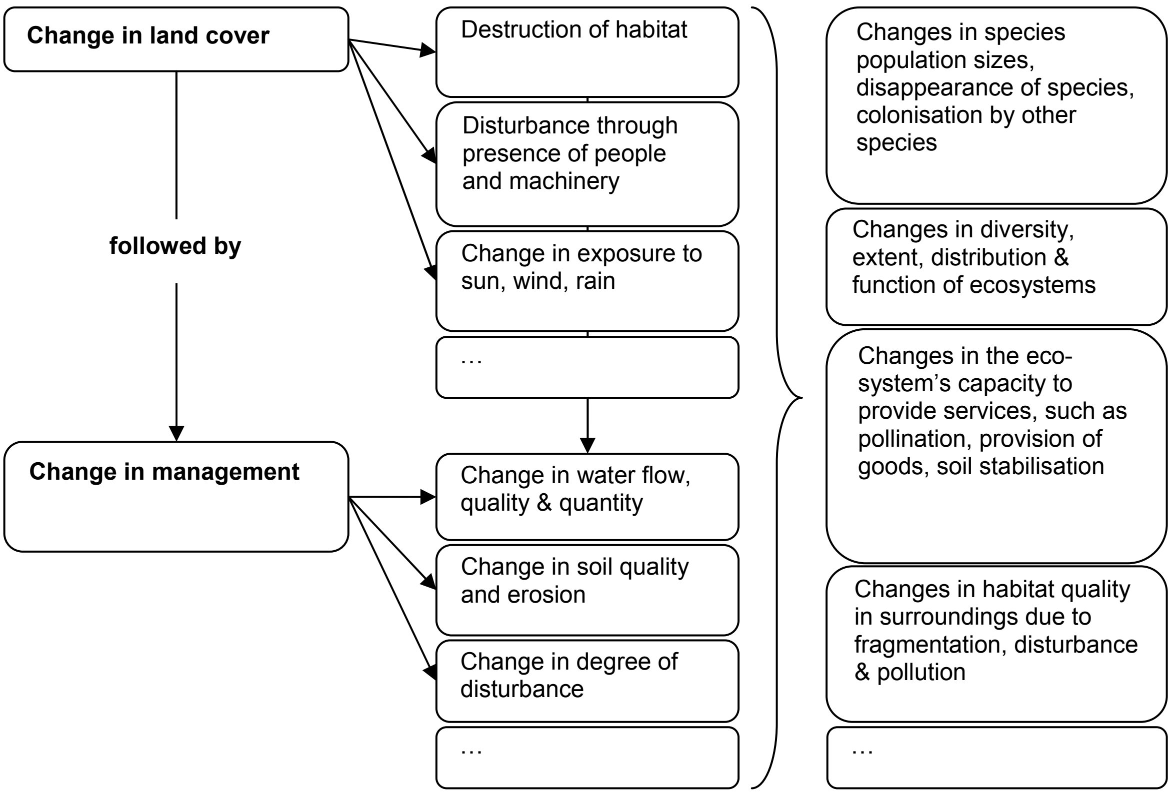

Figure 3. Land use change impacts on biodiversity. In conversion iLUC, both changes in land cover and changes in land management cause impacts on biodiversity. In intensification iLUC changes in land cover play a smaller role.

The implications of iLUC for biodiversity are essentially the same as those associated with direct land use change due to agricultural development. In conversion iLUC, both changes in land cover and changes in land management cause impacts on biodiversity, whereas for most cases of intensification iLUC, changes in land management are likely to be the most important factor. While intensification is mostly associated with negative effects on biodiversity due to increasing fertiliser and pesticide input, conversion iLUC can also have positive effects. For example, if the area converted for agriculture was previously degraded land, appropriate cultivation measures could enhance the quality of the soil and the vegetation structure, and therefore habitat quality could be enhanced (Tilman *et al* 2009).

In both cases, the effects on biodiversity are comprised of **on-site effects**, which occur, where the (indirect) land use change happens, and **off-site effects**, which occur in the surroundings as a consequence of the indirect land use change on a site (see figure). Other than the conversion from one ecosystem type to another (agricultural) type, on-site effects are likely to include the loss of species incapable of using the new agricultural system as habitat. Off-site effects, on the other hand, include: “contagious” effects of management practices, such as water use and drift of pesticides, herbicides and fertilizers and their effect on local biota; those resulting from changes in the landscape pattern, such as disruption of migration and foraging routes and isolation of remnant populations; and the effects of infrastructure development associated with conversion, such as installation of roads and power lines. Resulting edge effects on biodiversity include lower species population sizes and composition, invasion of disturbance-adapted species, lower humidity and air moisture and loss of living biomass, among many others (Broadbent *et al* 2008; Fletcher 2005).

Off-site impacts can be difficult to assess for several reasons. The area affected by off-site impacts is difficult to define and varies according to (a) land use and management, (b) the taxon or system of interest, and (c) the impact in question. For example, aerial pesticide drift may extend only a few metres, but once substances enter run-off, they may have impacts on freshwater organisms far removed from the site of application. Landscape effects are likely to act at scales of tens to hundreds of kilometres. Moreover, where changes are detected off-site, it may be difficult to trace them to particular areas of conversion or intensification (i.e. iLUC sites), especially in mosaic landscapes where other agricultural activities have similar impacts. Finally, on-site and off-site impacts differ in the timing of their development. Most on-site impacts follow immediately on conversion or intensification, but many off-site changes, such as changes in water quality and quantity, may happen more gradually. However, both on- and off-site effects can continue over years, and no attention has as yet been paid to the temporal dimension of iLUC effects on biodiversity.

In principle, the impacts of both conversion iLUC and intensification iLUC include both on-site and off-site effects in all areas along the ‘conversion chain’.

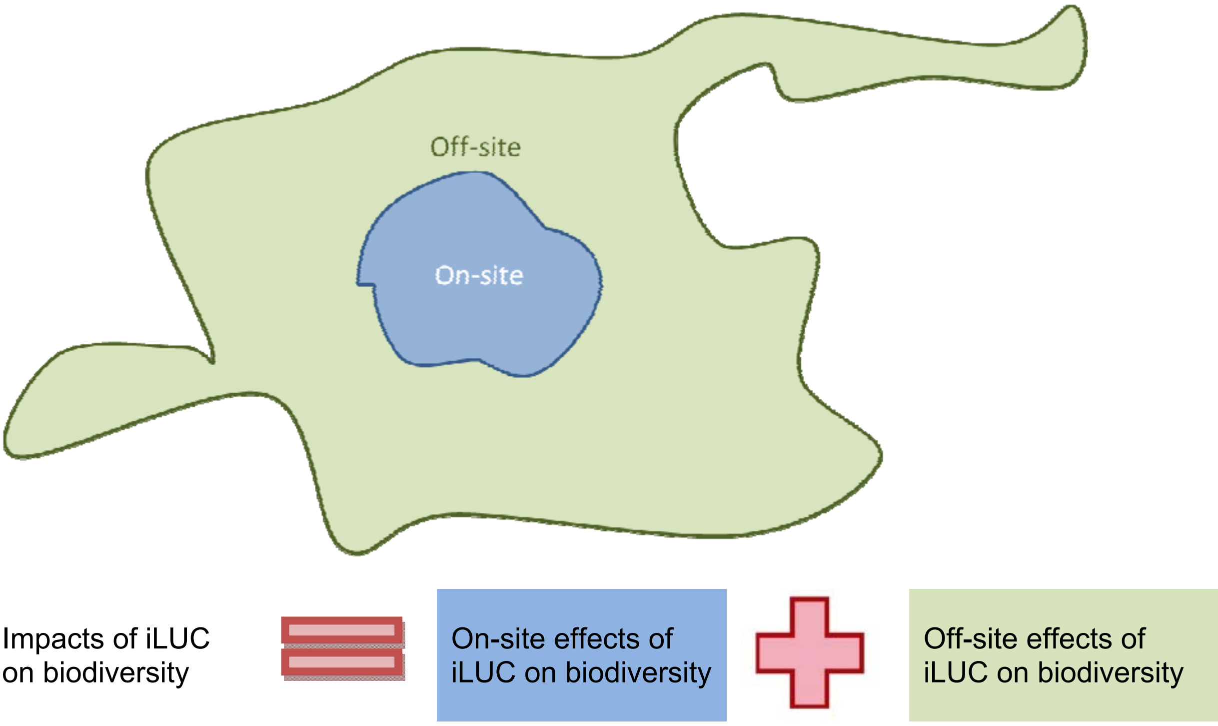

Figure 4. iLUC effects on biodiversity occur on-site and off-site.

Both on-site and off-site effects can include changes in species and their populations, changes in the extent and condition of ecosystems and changes in the composition of species and habitats (see Figure 3). However, on-site effects are likely to be more intense, as biodiversity in these areas is affected by land use in the first place and the cultivation methods and intensity of cultivation established subsequently. In contrast, biodiversity off-site is affected by the consequences of land use change and cultivation on-site but not by land use change per se.

In general, biodiversity impacts of iLUC depend on the form of iLUC and on where iLUC happens. While intensification is mostly associated with negative effects on biodiversity due to increasing fertiliser and pesticide input, conversion iLUC can also have positive effects. For example, if the area converted to agriculture was previously degraded land, the quality of the soil and therefore habitat quality could be enhanced through appropriate cultivation measures (Tilman *et al* 2009).

Recently attempts have been made to use models to assess the biodiversity impacts of iLUC from biofuel production (Hellmann & Verburg in press, Lapola *et al* 2010, van Oorschot *et al* 2010). One approach involves identifying hotspots of biodiversity impacts by overlaying spatially explicit information on the location of LUC derived from models with previously identified areas of biodiversity importance. The other approach involves more fully integrated biodiversity modelling.

Hellmann & Verburg (in press) intersected their spatially explicit model outputs for LUC due to biofuel production with semi natural vegetation, forest and high nature value farmland areas to identify hotspots of biodiversity impacts. They showed that 8.81-15.36 Mha and 0.96-1.49 Mha of semi natural vegetation and high nature value farmland respectively will be directly converted to biofuel production areas within the EU, whereas forest areas will increase in extent. The only study which models iLUC explicitly for Brazil at fine spatial resolution (*c.* 8km), Lapola *et al* (2010) showed that iLUC will cause expansion of rangeland into forest (12.2 Mha) and other native habitats (4.6 Mha). These two studies highlight the potential to assess the impacts of biofuel production on biodiversity in terms of ecosystem/habitat loss. Building on these model outputs, the impacts of such ecosystem/habitat loss on species could be assessed by incorporating information on species distributions and their sensitivity to habitat change (see below).

Van Oorschot *et al* 2010 evaluated the indirect effects of biofuel production on biodiversity in terms of “compensation period” using the GLOBIO model. The “compensation period” aims to capture both, biodiversity loss from land use change and avoided biodiversity loss because of avoided long-term climate change. The “compensation period” aims to be analogous to the carbon debt concept proposed by Searchinger *et al* (2008). The GLOBIO model assesses biodiversity impacts in terms of mean species abundance (MSA), an index of the intactness of an area. The GLOBIO model considers five pressures on biodiversity, land use change, fragmentation, nitrogen deposition, infrastructure and climate change, which are derived from the IMAGE integrated assessment model. However, for the analyses of iLUC only land use change impacts were assessed and intensification iLUC impacts, such as nitrogen fertilisers and pesticides, ignored. Depending on assumptions about the amount of iLUC in relation to direct land use change, the energy harvested per ha, and the greenhouse gas savings, the compensation periods can be several hundred years. Although the “compensation period” offers a tempting approach to combine positive and negative impacts of iLUC, it is not directly transferable from carbon to biodiversity. Unlike a carbon debt, a biodiversity debt cannot easily be “paid back” as extinct species cannot be brought back.

Existing efforts to model iLUC impacts on biodiversity are limited to conversion iLUC impacts. Even here models only include the direct impacts of conversion but not the subsequent impact of cultivation (e.g. the regular ploughing of land). Currently there are no models that consider the full impacts of iLUC, e.g. intensification iLUC, off-site impacts, and impacts along the conversion chain.

4 Measuring impacts of iLUC on biodiversity: framework for an ideal world

The complexities of iLUC and its implications for biodiversity highlighted in previous sections make the assessment of iLUC impacts on biodiversity extremely challenging, and have impeded the development of safeguards that might limit them.

Here we outline an analysis, based on the concepts set out above, of the information that would in theory be required to perform a thorough assessment iLUC impacts on biodiversity. We then examine the degree to which currently available information can meet these needs and identify information gaps. Subsequently, we propose some simplifying assumptions to enable the quantification of these impacts in the real world.

In principle, assessing the impacts of iLUC occurring at any given location would proceed through a framework consisting of five main steps (Figure 5):

1. Gather key information about the iLUC site;
2. Identify the biodiversity of the site;
3. Identify the on-site and off-site effects of iLUC;
4. Identify the impacts of these effects on components of biodiversity;
5. Assess overall biodiversity impact.

In an ideal world, one would be able to identify specific locations where conversion and intensification iLUC occur, and on-site and off-site impacts of iLUC on biodiversity would be assessed for each site that was affected. This would include all the areas along the 'conversion chain' and the areas surrounding each of them and all areas where intensification was caused through the expansion of biofuel production, as well as their surroundings. The framework thus represents the needs for assessment in relation to each iLUC site and addresses both on-site and off-site effects.

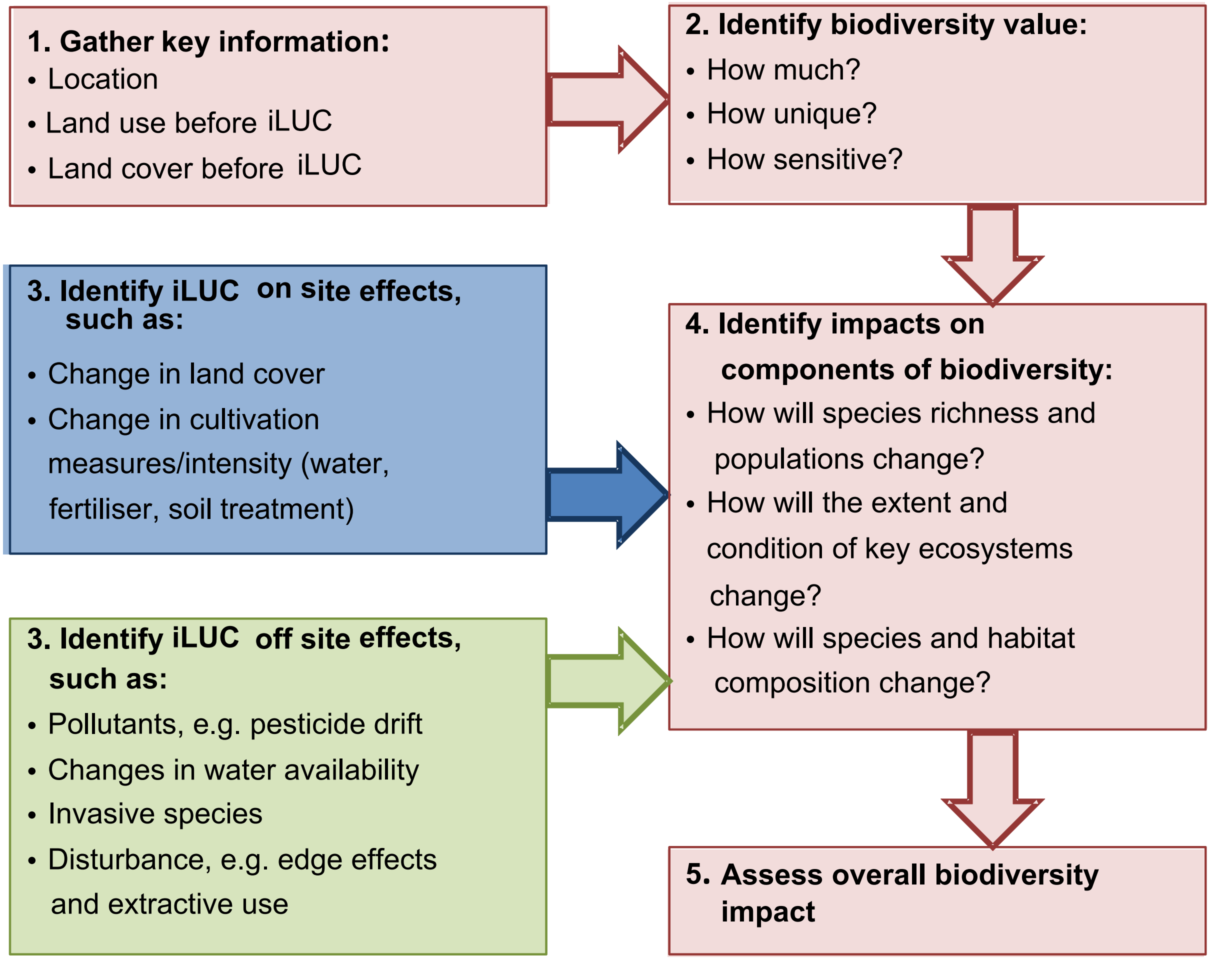

Figure 5. An 'ideal world framework' for assessing iLUC implications on biodiversity.
In the **first step**, key information about the location of the site as well as land use and land cover before iLUC needs to be gathered. The information on the location of the site needs to be spatially explicit, i.e. including the coordinates and boundaries of the affected site.

In order to enable the identification of the biodiversity value of the affected site in the next step, information is also needed on both the land use and the land cover of the site before bioenergy induced iLUC. Both are important because official designation of use may not reflect the biodiversity value of the land. For example, temporarily unstocked forest can be shown as forest on a land use map, even though the land may be without a single tree[1]. The implications for biodiversity of converting temporarily unstocked forest into agricultural land are much smaller than if the land is actually forested. Conversely, land use information is needed to supplement land cover maps, where certain categories that affect biodiversity value, such as managed and unmanaged grassland may not be differentiated.

In the **second step**, the biodiversity of the site before the land use change needs to be assessed. For a comprehensive picture of a site's biodiversity, information is required on, for example:

- Species richness and diversity of ecosystems, e.g. species numbers and numbers of different ecosystems;
- The uniqueness of the biodiversity, e.g. in terms of restricted range species or rare ecosystems;

[1] This is due to the current forest definition applied in the UNFCCC negotiations, see FCCC/CP/2001/13/Add.1

- The vulnerability and sensitivity of its biodiversity, including how threatened its components are, and their tolerance to disturbance, changes in vegetation cover, etc.

The quantity of biodiversity, i.e. an estimate of species and ecosystem richness, is not considered sufficient to estimate the biodiversity value of the site. Including the uniqueness of biodiversity is of particular importance because local impacts on restricted-range species and rare ecosystems can have national, regional or even global conservation implications. These implications are even stronger where components of biodiversity are threatened, and information on the sensitivity of species and ecosystems to change helps to determine the effect that iLUC will have.

Additional information on the biodiversity value of a site prior to iLUC could be derived from the broader landscape pattern. For example, the affected site may be located in a mosaic landscape with a mixture of agricultural areas in the surroundings or in an area as yet 'untouched'. Such differences in the landscape contribute to the biodiversity at the site itself, but will also influence edge effects and opportunities for species to move to more suitable areas following the land use change. The spatial information on land use and land cover from step 1 could be used to provide a better picture of the landscape pattern. This information could then be used together with data on the sensitivity of species to better determine their reaction to change in the next step.

In summary, the overall information needed to assess the biodiversity value of a site prior to iLUC impacts includes the following, with the addition of information on the landscape pattern:

	Number	Threatened	Endemic
Ecosystems			/
• Thereof sensitive			/
Birds			
• Thereof sensitive			
Mammals			
• Thereof sensitive			
Vascular Plants			
• Thereof sensitive			
… to be continued for other taxonomic groups			

In **step 3**, the nature of the iLUC-induced changes at the site needs to be determined. This includes identifying changes in the land cover (for conversion iLUC) as well as the agricultural practices established subsequently (including intensification). Different agricultural practices can differ significantly in their impacts on different components of biodiversity. For example, such practices as low- or no-till agriculture, integrated pest and nutrient management, agroforestry/intercropping and low impact harvesting, have very different implications for biodiversity from intensive, high input cultivation (e.g. Gemmel 2001). Identification of agricultural practices is also necessary to help identify off-site impacts, such as movement of fertilisers and/or pesticides through various pathways. For example, highly soluble forms of nitrogen and some pesticides are likely to infiltrate downwards towards the water table, while other substances may move significant distances through surface run-off, volatilization, and wind movement. It is also important to identify the landscape changes caused by the iLUC, such as fragmentation and creation of new edges.

In **step 4**, the combined information from the previous two steps is used to determine the actual iLUC implications for components of biodiversity. There is a growing body of literature on the impact of different agricultural or forestry systems on biodiversity. For example, it has been found that biodiversity in poplar plantations is higher than in row crops, lower than in forests, but about equal to grasslands (Smeets *et al* 2005). However, such studies are often limited to species richness and compare the number of species in different systems rather than considering biodiversity more comprehensively and comparing the overall status before and after the land use change. We therefore consider the following questions as important for assessing the biodiversity impact on- and off-site:

- How do species richness and populations change?
- How do extent and condition of key ecosystems change?
- How does community composition change?

The first question not only addresses the impact on species numbers but also on population sizes. This is considered of particular importance for endangered and endemic species, where a change in population sizes is more likely to impact their overall conservation status. The second question aims at covering the same aspects for each of the affected ecosystems. Here, the change in extent could be included as percentage change and expressed in relation to the national extent of, and background rates of change in the ecosystem. Especially for critically endangered ecosystems this information would be valuable to highlight conservation needs. The third question tries to capture the magnitude of the combined changes in species abundance caused by iLUC and their implications for whole communities. Even where iLUC-induced changes in the numbers of species or extent of ecosystems are small there may be appreciable changes in community composition.

Ideally, these changes would be estimated based on an impact matrix (
Figure 6) for each species summarising the effects of conversion from one land use to another. Effects could be expressed in terms of changes in species' abundance or population viability. Such matrices could be used as look up tables to derive the biodiversity impact of iLUC on species at a site.

Conversion
to from

	Forest	Grassland	Coffee	Carrots	Potatoes	Tomatoes
Forest	-					
Grassland		-				
Coffee			-			
Carrots				-		
Potatoes					-	
Tomatoes						-

Figure 6. A species impact matrix, where each cell would contain an estimate for the change in abundance of a species after conversion from one land use to another.

In **step 5**, the overall impact on biodiversity is to be assessed by aggregating and summarising the information compiled in previous steps to provide an overview of biodiversity impacts. The following table provides an example for how this can be done by ecosystem type affected by iLUC (Table 2). Further aggregation (e.g. a summary of biodiversity impacts per site affected by iLUC) is not recommended as the effects on different ecosystems and their species composition and functions can vary significantly and important information on biodiversity impacts could get lost by their aggregation. Separately from this table, information on the number of species whose population sizes have decreased, increased or remained unchanged could be provided. It is crucial to note that even in an ideal world with unlimited information, biodiversity impacts of land use change cannot be summarised using a single metric.

Table 2. Example for summarising overall impact of biodiversity by ecosystem affected.

Site 1: [Ecosystem and % of site covered by this ecosystem]			
	Before iLUC	**After iLUC**	**Δ**
Ecosystem extent	X hectares	Y hectares	Before - after
Ecosystem condition	Scale from 1-5	Scale from 1-5	Before - after
Number of species	By taxonomic group	By taxonomic group	Before –after by taxonomic group
Number of threatened species	By taxonomic group	By taxonomic group	Before –after by taxonomic group
Number of endemic species	By taxonomic group	By taxonomic group	Before –after by taxonomic group
Composition			Similarity index
Landscape fragmentation/ connectivity			

5 Moving the framework towards practice

The presented ideal world framework, as the name suggests, cannot be directly transferred into practice. This is partly due to the fact that iLUC cannot be traced back because of the complexities discussed earlier, e.g. because it can be displaced into other countries, occur with time lag and be distributed through global trading (Fritsche & Wiegmann 2011). However, there are also some data gaps that make practical implementation of the framework impossible. In the following, an overview on available data that may be useful for assessing the impacts of iLUC on biodiversity is provided, main data gaps summarised and potential simplifications to the framework will be suggested and discussed.

5.1 Availability of relevant data and data gaps

For step one of the framework, spatial information is needed on **land use, land cover and ecological characteristics** of the affected and surrounding areas.

When searching for land use and land cover information, it will be important to consider the age of the data as well as their resolution to make sure the data is suitable for this purpose. In many cases, governmental departments involved in spatial planning processes are holding such information. Where this is not available, existing land cover and use information derived from remote sensing could be explored, e.g. from USGS Landsat.

For an indication of ecological characteristics of the affected site and its surroundings, such spatial information could then be combined with WWF's terrestrial and freshwater ecoregions (Abell *et al* 2008; Olson *et al* 2001). The GIS shapefiles for both the terrestrial and freshwater ecoregions are available for download (http://www.worldwildlife.org/science/data/item6373.html and http://www.feow.org/downloads/GIS1.1.zip).

For a description of the ecoregions, WWF's WildFinder (http://gis.wwfus.org/wildfinder/), an online interface to search for places and ecoregions that includes links to more detail about each ecoregion, including its geography, biodiversity and threats, can be used. The website for the freshwater ecoregions of the world (http://www.feow.org/) includes an indication for freshwater species richness and number of endemic freshwater species, which is of interest to the next step of the framework.

Ideally, information about the intensity of disturbance and cultivation methods at the site itself and in surrounding areas before the land use change should be compiled as well as part of this step. This would allow a more comprehensive assessment of biodiversity under different disturbance regimes as well as impacts on biodiversity in later steps. The degree of disturbance can partly be derived from the land cover and land use data, but additional information on the cultivation methods of the land may still be needed. This information may be difficult to compile.

For step two of the framework, information on **biodiversity richness, uniqueness, vulnerability and sensitivity** is needed.

At best, data on **biodiversity richness** for a site affected by iLUC and its surroundings would come from site-level assessments and consist of the number of species occurring at the site divided by taxonomic group (e.g. mammals, birds, vascular plants etc.). Such assessments exist for some areas, e.g. Conservation International's Rapid Assessment Program provides information from assessments in almost 60 areas, including on endemic and threatened species occurring in these places (https://learning.conservation.org/biosurvey/RAP/Pages/ Results.aspx). However, in most cases such data will not exist. One of the major sources for information on species richness is the International Union for Conservation of Nature (IUCN). Available species richness information includes:

- Global geographic patterns of mammal species richness: http://www.iucnredlist.org/initiatives/mammals/analysis/geographic-patterns
- Global geographic patterns of amphibian species richness: http://www.iucnredlist.org/initiatives/amphibians/analysis/geographic-patterns
- Sub-global geographic patterns of the richness of other taxa, such as for Pan-Africa: http://www.iucnredlist.org/initiatives/freshwater/panafrica/geographic

BirdLife International is currently finalising global maps of bird species richness. Concerning vascular plants, Barthlott *et al* (2005) have produced a global map of vascular plant species richness. A number of sub-global initiatives have produced richness maps as well, such as NatureServe for Latin America (http://www.natureserve.org) and the Columbia University, SEDAC and NatureServe jointly for the whole of the Americas (http://sedac.ciesin.columbia.edu/species/).

For the European context, the European Commission hosts information on geographic patterns of European species richness by taxonomic group on their website (http://ec.europa.eu/environment/nature/conservation/species/redlist/). Species richness maps can be viewed for mammals, amphibians, reptiles, freshwater fishes, butterflies, dragonflies, saproxylic beetles and molluscs and an assessment of vascular plant species richness is underway. In addition, the European Environment Agency provides access to coarse maps on richness of reptiles, breeding birds, mammals and amphibians in Europe. However, the resolution of the maps is very limited.

Species richness at a specific site may be lower than indicated by the coarse scale global sources mentioned above. Where information on the degree of disturbance and intensity of cultivation before the indirect land use change due to biofuels exists, the species richness and ecosystem diversity of the site may need adjusting. A literature review may help understand how the current cultivation or disturbance regime may have impacted the biodiversity before the land use change.

Concerning the **uniqueness and vulnerability of biodiversity** there are a number of global and regional approaches for prioritising areas of particular importance for biodiversity and conservation, e.g. according to the number of endemic or threatened species occurring in these places. Examples include:

- Important Plant Areas (Plantlife 2010);
- Important Bird Areas (BirdLife International 2011);
- Alliance for Zero Extinction Sites (Alliance for Zero Extinction 2010);
- WWF’s global 200 ecoregions (Olson & Dinerstein 2002; Olson & Dinerstein 1998);
- Global patterns of endemism richness for vascular plants, vertebrates, amphibians, reptiles, birds and mammals (Kier *et al* 2009);
- Ecological gap analyses (CBD 2007; Rodrigues *et al* 2004);
- Conservation International’s Biodiversity Hotspots (Mittermeier *et al* 2004; Myers *et al* 2000)
- Threatened Ecosystems of the World (Rodriguez *et al* 2011);
- High Nature Value Farmland in Europe (Paracchini *et al* 2008);

In addition, the IUCN Red List on Threatened Species provides valuable information on the threat status of species, the distribution of threatened species, pressures on them and their characteristics. It may also be of interest to consider whether the area and surrounding areas are included in a protected area or not. The World Database on Protected Areas provides spatial information on protected areas globally (http://www.wdpa.org/). For concise information about a large number of approaches for biodiversity prioritisation, see the "a-z of areas of biodiversity importance" at http://www.biodiversitya-z.org/.

Information on the vulnerability of biodiversity components exists but is scattered. GLOBIO, the Global Biodiversity Model for policy support (http://www.globio.info/), contains some information on the vulnerability of species to land use change but is largely based on information from tropical countries. For the European context, BioScore, a European biodiversity impact assessment tool, could be a valuable source of information (http://www.bioscore.eu/, see Delbaere *et al* 2009). The tool provides indicator values for the ecological preferences of more than 1,000 species of birds, mammals, amphibians, reptiles, fish, butterflies, dragonflies, aquatic macro-invertebrates and vascular plants.

For step 3 of the framework, information is needed on **what exactly is happening at the site affected by iLUC** and its surroundings. At best, this information would come from field observation. Apart from information on the crop established after conversion (not applicable to intensification iLUC), information is needed on the established cultivation method (both for conversion and intensification iLUC). For conversion iLUC it may be possible to derive this information from remotely sensed data but for intensification iLUC this will not be easily possible.

In step 4 of the framework, the **actual impact on biodiversity** at the site and surrounding areas is to be assessed. The previously gathered information on the vulnerability of different biodiversity components to change should be used. In addition, a large body of scientific literature exists on the impacts of different disturbance regimes on biodiversity and ecosystem services (e.g. Broadbent *et al* 2008; Fletcher 2005; Hermy & Verheyen 2007; Laurance *et al* 2007; Liira *et al* 2008; Norris *et al* 2010). Impacts will differ from site to site and vary between biodiversity components. We are not aware of a database on the impacts of conversion and intensification on different biodiversity components in different locations. Consequently, this step requires site-specific assessments based on the information compiled in previous steps.

In an ideal world, this biodiversity impact assessment would be based on site-scale information on biodiversity components and cultivation regimes before and after the land use change on-site as well as off-site. However, in most cases, site-scale information will not be available and field surveys are mostly not feasible due to the time and budget required. Despite the limitations of global and regional data sets (e.g. referring to their resolution and the limited ability to consider site-specific contexts), existing information allows assessing the situation at the site before the land use change up to a certain extent. However, for the assessment of the changes in biodiversity components after the land use change, information is much more scarce and scattered and global datasets cannot be used. Site-scale information on the cultivation and disturbance regimes on-site and off-site after the land use change will be even more important for this step of the framework. We conclude that major information gaps exist in the following areas:

- Cultivation methods at site before and after the change;
- Cultivation methods/degree of disturbance in the surroundings before and after the change;
- Boundaries of off-site impacts;
- The vulnerability of biodiversity components and ecosystem services to change.

For these assessments, knowledge of the exact location of iLUC is crucial but mostly unknown. This lack of knowledge probably represents the major information gap.

5.2 Potential simplifying assumptions to increase the feasibility for implementation of the framework

Based on the previous sections we consider it necessary to identify potential simplifying assumptions that can help implementing the framework. These assumptions will reduce the overall comprehensiveness of the assessment but make implementation feasible:

5.2.1 Spatially explicit model results can be used to identify where iLUC is happening now or in the future

Taken that the exact location of iLUC will remain very difficult, if not impossible, to identify, it will be necessary to use spatially explicit model results about where iLUC is happening or going to happen in the future in order to assess its impacts on biodiversity. Using such spatially explicit model results together with some of the datasets on biodiversity identified above will allow for a coarse assessment of the biodiversity impacts of conversion iLUC. However, so far these models do not consider intensification iLUC. As discussions on iLUC evolve, it will be interesting to see whether this distinction between the two forms of iLUC is being recognised and will be taken into consideration in future developments of spatially explicit models.

5.2.2 Replacement of one crop by another crop has no biodiversity implications

By assuming that there are no biodiversity impacts of replacing one crop with another the sites potentially affected by conversion iLUC along the conversion chain can be ignored. Although it is known that some replacements can cause considerable changes in biodiversity the assumption is considered necessary to implement the framework as currently the location of affected sites along the conversion chain cannot be identified.

5.2.3 Assume off-site effects occur at a constant width around the affected site

A major challenge in assessing the off-site impacts of iLUC on biodiversity is that their boundaries are unknown, i.e. it is unknown how far they expand into areas around the affected sites. One way to address this challenge is to assume a fixed width within which biodiversity is considered to be affected around the iLUC site. This may lead to over- and/or underestimates of the size of the area affected depending on off-site effects considered, but this is considered a necessary simplification to address off-site impacts.

5.2.4 Not all species need to be assessed

It will always remain challenging to assess the impacts of land use change on all components of biodiversity. However, some taxonomic groups are particularly well researched, such as birds and mammals, and some can be considered as indicator species. Hence it is suggested to reduce the scope of the assessment to well-known taxonomic groups.

5.2.5 Quantification and aggregation of impacts is not always necessary

Conducting fully quantitative and aggregated assessments of biodiversity impacts of iLUC is hardly ever possible. Apart from the difficulties related to identifying the exact location of iLUC, there is no single unit for biodiversity that can be measured. Moreover, aggregating responses of biodiversity to change will oversimplify variable and potentially conflicting responses. Descriptive, qualitative assessments of changes in biodiversity and ecosystem services following iLUC can be important tools to improve our understanding and communicate the overall impacts of biofuels on biodiversity. This is in line with the precautionary principle and represents the best way forward in absence of quantitative evidence.

6 Conclusions

Inference from production statistics and modelling studies suggest that iLUC has occurred, is likely to increase with increasing demand for biofuels and needs to be considered when assessing the biodiversity impacts of biofuels. Although additional GHG emissions from iLUC have been considered by various studies (e.g. Searchinger *et al* 2008), biodiversity impacts of iLUC have so far only been assessed by a limited number of modelling studies and empirical assessments are still lacking.

Biodiversity implications of iLUC may differ spatially and temporally from GHG implications of iLUC. Additional GHG emissions result purely from land conversion and intensification on-site whereas biodiversity can be affected both on- and off-site. Implications for GHG emissions are not necessarily correlated with biodiversity implications, i.e. in areas where GHG emission impacts are small, impacts on biodiversity may be large and vice versa. Furthermore, time lags for biodiversity impacts, e.g. extinction lags, are likely to be longer than for GHG emissions. While calculating the carbon debt from additional GHG emissions due to iLUC is possible, the calculation of a biodiversity debt is not valid. Firstly, there is no single metric for biodiversity and as a consequence biodiversity impacts cannot be aggregated. Secondly, not all ecosystem and biodiversity loss can be recovered, in particular species extinction are irreversible. Furthermore, biodiversity varies across space hence no single factor for biodiversity impacts can be developed (unlike the GHG factor as all carbon is equal), even if likelihood and location of iLUC are known.

Assessing biodiversity impacts requires spatial information about where iLUC is happening and the biodiversity value of the site prior to land use change. Existing studies reviewed here only consider conversion iLUC but usually ignore both, steps along the conversion chain and intensification iLUC. Even when information on the location of iLUC is available, this is usually restricted to the site at the end of the conversion chain, while sites along the chain are often missed out. As a consequence, any assessment of biodiversity impacts is going to be conservative. While intensification is considered a solution to iLUC it can represent a form of iLUC in itself. Although intensification may have less biodiversity implications compared to conversion of land not currently under agricultural production, there are biodiversity implications, which should not be overlooked.

Sustainability standards and criteria for first generation biofuel crops, such as those incorporated in the European Union's Renewable Energy Directive (European Union 2009), aim at preventing biofuel production encroaching on areas of importance for biodiversity and ecosystem services. Hence they represent a mechanism to control where conversion for biofuel production will take place in the future. However, because of its complexities, there are currently no standards or criteria that can prevent iLUC from happening. This presents a gap in the sustainability standards: By banning biofuel crops from certain areas, their cultivation on existing agricultural land is encouraged, thereby forcing the food crops and feedstocks previously grown on this land to move elsewhere (iLUC) (Searchinger *et al* 2008). However, these food crops or feedstock can enter the areas that biofuel crops are banned from. The logical conclusion is that sustainability standards and criteria for biofuel production will not be able to ensure sustainability without a criterion on iLUC. Furthermore, biodiversity impacts from intensification iLUC are not considered in any policies and standards. There is an urgent need to continue developing biodiversity safeguards for biofuel production that take the potential biodiversity implications of iLUC in all its dimensions into consideration.

Given the likelihood that estimates of iLUC impacts on biodiversity will be conservative and the lack of effective safeguards in this regard, it is essential to take a precautionary approach in developing and sourcing biofuels.

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8 References

ABELL, R., THIEME, M.L., REVENGA, C., BRYER, M., KOTTELAT, M., BOGUTSKAYA, N., COAD, B., MANDRAK, N., BALDERAS, S.C., BUSSING, W., STIASSNY, M.L.J., SKELTON, P., ALLEN, G.R., UNMACK, P., NASEKA, A., NG, R., SINDORF, N., ROBERTSON, J., ARMIJO, E., HIGGINS, J.V., HEIBEL, T.J., WIKRAMANAYAKE, E., OLSON, D., LÓPEZ, H.L., REIS, R.E., LUNDBERG, J.G., SABAJ PÉREZ, M.H. & PETRY, P. 2008. Freshwater Ecoregions of the World: A New Map of Biogeographic Units for Freshwater Biodiversity Conservation. *Bioscience,* **58**, 403-414.

ADEN, A. 2007. Water usage for current and future ethanol production. *Southwest Hydrology,* September/October, 22-23.

ALLIANCE FOR ZERO EXTINCTION. 2010. *Alliance for Zero Extinction. Pinpointing and Preventing Imminent Extinctions* [http://www.zeroextinction.org/] Accessed 4.8.2010.

BANSE, M., VAN MEIJL, H., TABEAU, A., WOLTJER, G., HELLMANN, F. & VERBURG, P.H. (in press). Impact of EU biofuel policies on world agricultural production and land use. *Biomass and Bioenergy* 10.1016/j.biombioe.2010.09.001.

BARTHLOTT, W., KIER, G., KREFT, H., KÜPER, W., RAFIQPOOR, D. & MUTKE, J. 2005. *Global biodiversity: species numbers of vascular plants*. Nees Institute for Biodiversity of Plants, University of Bonn, Bonn, Germany.

BAUEN, A., CHUDZIAK, C., VAD, K. & WATSON, P. 2010. A causal descriptive approach to modelling the GHG emissions associated with the indirect land use impacts of biofuels. A study for the UK department for transport. E4tech, London, UK.

BIRDLIFE INTERNATIONAL. 2011. *Important Bird Areas. BirdLife International.* Cambridge, UK. [http://www.birdlife.org/action/science/sites/index.html] Accessed 14.4.2011.

BORYSOV, S. 2011. Perspectives of Biofuel Market Development in Ukraine. [http://www.iamo.de/uploads/media/5a.4_Borysov_CD.pdf] Accessed 30.3.2011.

BOWYER, C. 2010. *Anticipated indirect land use change associated with expanded use of biofuels and bioliquids in the EU - an analysis of the national renewable energy action plans*. IEEP - 24 pages.

BROADBENT, E.N., ASNER, G.P., KELLER, M., KNAPP, D.E., OLIVEIRA, P.J.C. & SILVA, J.N. 2008. Forest fragmentation and edge effects from deforestation and selective logging in the Brazilian Amazon. *Biological Conservation,* **141**, 1745-1757.

CBD. 2007. *Ecological Gap Analysis.* [http://www.cbd.int/protected-old/gap.shtml] Accessed 14.4.2011.

COMMISSION OF THE EUROPEAN COMMUNITIES. 2009. *Directive 2009/28/EC of the European Parliament and of the Council of 23 April 2009 on the promotion of the use of energy from renewable sources and amending and subsequently repealing Directives 2001/77/EC and 2003/30/EC*. [http://eur-lex.europa.eu/LexUriServ/LexUriServ.do?uri=OJ:L:2009:140:0016:0062:EN:PDF] Accessed 16.2.2011.

CROEZEN, H.J., BERGSMA, G.C., OTTEN, M.B.J. & VAN VALKENGOED, M.P.J. 2010. *Biofuels: indirect land use change and climate impact*. CE Delft, 1-62.

DELBAERE, B., NIETO SERRADILLA, A. & SNETHLAGE, M. (eds.). 2009. *BioScore: A tool to assess the impacts of European Community policies on Europe's biodiversity*. ECNC, Tilburg, the Netherlands.

EDWARDS, R., MULLIGAN, D. & MARELLI, L. 2010. *Indirect Land Use Change from increased biofuels demand. Comparison of models and results for marginal biofuels production from different feedstocks*. European Union, Joint Research Centre, Institute for Energy, Publications Office of the European Union, Luxembourg.

ELBERSEN, W., WIERSINGA, R. & WAARTS, Y. 2009. *Market Scan Bioenergy Ukraine.* Report for the Dutch Ministry of Agriculture, Nature and Food Quality. Agrotechnology & Food Innovations B.V., Wageningen, The Netherlands.

EUROPEAN UNION. 2009. The promotion of the use of energy from renewable sources. Directive 2009/28/EC L140. *Official Journal of the European Union* 52.

FARGIONE, J.E., COOPER, T.R., FLASPOHLER, D.J., HILL, J., LEHMAN, C., MCCOY, T., MCLEOD, S., NELSON, E.J., OBERHAUSER, K.S. & TILMAN, D. 2009. Bioenergy and Wildlife: Threats and Opportunities for Grassland Conservation. *Bioscience,* **59**, 767-777.

FARGIONE, J.E., PLEVIN, R.J. & HILL, J.D. 2010. The Ecological Impact of Biofuels. *Annual Review of Ecology, Evolution, and Systematics,* **41**, 351-377.

FLETCHER, R.J. 2005. Multiple edge effects and their implications in fragmented landscapes. *Journal of Animal Ecology,* **74**, 342-352.

FRITSCHE, U.R., HENNENBERG, K. & HÜNECKE, K. 2010. The "iLUC Factor" as a Means to Hedge Risks of GHG Emissions from Indirect Land Use Change. Working Paper: Öko-Institut, Darmstadt Office, Darmstadt, Germany.

FRITSCHE, U.R. & WIEGMANN, K. 2011. *Indirect Land Use Change and Biofuels*. Study: European Parliament, Brussels, Belgium.

GALLAGHER, E. 2008. *The Gallagher Review of the indirect effects of biofuels production.* Renewable Fuels Agency, UK.

GEMMEL, B. 2001. *Managing Agricultural Resources for Biodiversity Conservation: A guide to best practices*. Environment Liaison Centre International, Nairobi, Kenya. Produced with the support of the UNEP/UNDP GEF Biodiversity Planning Support Program, available at http://www.ukabc.org/agbioguide.pdf

HELLMANN, F. & VERBURG, P.H. (in press). Impact assessment of the European biofuel directive on land use and biodiversity. *Journal of Environmental Management,* **91**, 1389-1396.

HERMY, M. & VERHEYEN, K. 2007. Legacies of the past in the present-day forest biodiversity: a review of past land-use effects on forest plant species composition and diversity. *Ecological Research,* **22**, 361-371.

IEA. 2006. World Energy Outlook 2006. OECD/IEA, Paris, France.

KIER, G., KREFT, H., LEE, T.M., JETZ, W., IBISCH, P.L., NOWICKI, C., MUTKE, J. & BARTHLOTT, W. 2009. A global assessment of endemism and species richness across island and mainland regions. *PNAS Early Edition,* **106**, 9322-9327.

KYRYZYUK, S. 2010. *Model-based risk-adjusted planning for sustainable agriculture under agricultural trade liberalization: Ukrainian case study*. Interim report IR-10-016. International Institute for Applied Systems Analysis (IIASA), Laxenburg, Austria.

LAMBIN, E. & MEYFROIDT, P. 2011. Global land use change, economic globalization, and the looming land scarcity. *Proceedings of the National Academy of Sciences of the United States of America*.

LAPOLA, D.M., SCHALDACH, R., ALCAMO, J., BONDEAU, A., KOCH, J., KOELKING, C. & PRIESS, J.A. 2010. Indirect land-use changes can overcome carbon savings from biofuels in Brazil. *Proceedings of the National Academy of Sciences,* **107**, 3388-3393.

LAURANCE, W.F., NASCIMENTO, H.E.M., LAURANCE, S.G., ANDRADE, A., EWERS, R.M., HARMS, K.E., LUIZÃO, R.C.C. & RIBEIRO, J.E. 2007. Habitat Fragmentation, Variable Edge Effects, and the Landscape-Divergence Hypothesis. *Plos One* 2, e1017.

LIIRA, J., SCHMIDT, T., AAVIK, T., ARENS, P., AUGENSTEIN, I., BAILEY, D., BILLETER, R., BUKÁCEK, R., BUREL, F., DE BLUST, G., DE COCK, R., DIRKSEN, J., EDWARDS, P.J., HAMERSKÝ, R., HERZOG, F., KLOTZ, S., KÜHN, I., LE COEUR, D., MIKLOVÁ, P., ROUBALOVA, M., SCHWEIGER, O., SMULDERS, M.J.M., VAN WINGERDEN, W.K.R.E., BUGTER, R. & ZOBEL, M. 2008. Plant functional group composition and large-scale species richness in European agricultural landscapes. *Journal of Vegetation Science,* **19**, 3-14.

LONDO, M., DEURWAARDER, E., LENSINK, S., FISCHER, G., PRIELER, S., VAN VELTHUISEN, H., DE WIT, M., FAAIJ, A., BERNDES, G., HANSSON, J. & DUER, H. 2007. *REFUEL: An EU Roadmap for Biofuels.* 15th European Biomass Conference & Exhibition, 7-11 May 2007, Berlin, Germany. 2982-2933.

MELILLO, J.M., REILLY, J.M., KICKLIGHTER, D.W., GURGEL, A.C., CRONIN, T.W., PALTSEV, S., FELZER, B.S., WANG, X., SOKOLOV, A.P. & SCHLOSSER, C.A. 2009. Indirect Emissions from Biofuels: How Important? *Science,* **326**, 1397-1399.

MITTERMEIER, R.A., ROBLES-GIL, P., HOFFMANN, M., PILGRIM, J.D., BROOKS, T., MITTERMEIER, C.G., LAMOREUX, J. & DA FONSECA, G.A.B. 2004. Hotspots Revisited: Earth's Biologically Richest and Most Endangered Terrestrial Ecoregions. CEMEX, Mexico City, Mexico.

MYERS, N., MITTERMEIER, R.A., MITTERMEIER, C.G., DA FONSECA, G.A.B. & KENT, J. 2000. Biodiversity hotspots for conservation priorities. *Nature,* **403**, 853-858.

NORRIS, K., ASASE, A., COLLEN, B., GOCKOWKSI, J., MASON, J., PHALAN, B. & WADE, A. 2010. Biodiversity in a forest-agriculture mosaic - The changing face of West African rainforests. *Biological Conservation,* **143**, 2341-2350.

OLSON, D.M. & DINERSTEIN, E. 2002. The Global 200: Priority Ecoregions for Global Conservation. *Annals of the Missouri Botanical Garden,* **89**, 199-224.

OLSON, D.M., DINERSTEIN, E., WIKRAMANAYAKE, E.D., BURGESS, N.D., POWELL, G.V.N., UNDERWOOD, E.C., D'AMICO, J.A., ITOUA, I., STRAND, H.E., MORRISON, J.C., LOUCKS, C.J., ALLNUTT, T., RICKETTS, T.H., KURA, Y., LAMOREUX, J.F., WETTENGEL, W.W., HEDAO, P. & KASSEM, K.R. 2001. Terrestrial Ecoregions of the World: a new map of life on Earth. *Bioscience,* **51**, 933-938.

OLSON, D.M. & DINERSTEIN, E. 1998. The Global 200: A Representation Approach to Conserving the Earth's Most Biologically Valuable Ecoregions. *Conservation Biology,* **12**, 502-515.

OVERMARS, K.P., STEHFEST, E., ROS, J.P.M. & PRINS, A.G. 2011. Indirect land use change emissions related to EU biofuel consumption: an analysis based on historical data. *Environmental Science & Policy,* **14**, 248-257.

PARACCHINI, M.L., PETERSEN, J.-E., HOOGEVEEN, Y., BAMPS, C., BURFIELD, I. & VAN SWAAY, C. 2008. *High Nature Value Farmland in Europe. An estimate of the distribution patterns on the basis of land cover and biodiversity data.* EUR 23480 EN - 2008. European Commission, Joint Research Centre, Institute for Environment and Sustainability, European Environment Agency, Office for Official Publications of the European Communities, Luxembourg.

PHALAN, B. 2009. The social and environmental impacts of biofuels in Asia: An overview. *Applied Energy,* **86**, S21-S29.

PLANTLIFE. 2010. *Important Plant Areas around the World. Target 5 of the CBD Global Strategy for Plant Conservation*. Plantlife International, Salisbury, UK.

PLEVIN, R.J., O'HARE, M., JONES, A.D., TORN, M.S. & GIBBS, H.K. 2010. Greenhouse Gas Emissions from Biofuels' Indirect Land Use Change Are Uncertain but May Be Much Greater than Previously Estimated. *Environmental Science & Technology,* null.

RODRIGUES, A.S.L., AKCAKAYA, H.R., ANDELMAN, S.J., BAKARR, M.I., BOITANI, L., BROOKS, T.M., CHANSON, J.S., FISHPOOL, L.D.C., DA FONSECA, G.A.B., GASTON, K.J., HOFFMANN, M., MARQUET, P.A., PILGRIM, J.D., PRESSEY, R.L., SCHIPPER, J., SECHREST, W., STUART, S.N., UNDERHILL, L.G., WALLER, R.W., WATTS, M.E.J. & YAN, X. 2004. Global Gap Analysis: Priority Regions for Expanding the Global Protected-Area Network. *Bioscience,* **54**, 1092-1100.

RODRIGUEZ, J.P., RODRÍGUEZ-CLARK, K.M., BAILLIE, J.E.M., ASH, N., BENSON, J., BOUCHER, T., BROWN, C., BURGESS, N.D., COLLEN, B., JENNINGS, M.D., KEITH, D.A., NICHOLSON, E., REVENGA, C., REYERS, B., ROUGET, M., SMITH, T., SPALDING, M., TABER, A., WALPOLE, M., ZAGER, I. & ZAMIN, T. 2011. Establishing IUCN Red List Criteria for Threatened Ecosystems. Establecimiento de Criterios para la Lista Roja de UICN de Ecosistemas Amenazados. *Conservation Biology,* **25**, 21-29.

SEARCHINGER, T., HEIMLICH, R., HOUGHTON, R.A., DONG, F.X., ELOBEID, A., FABIOSA, J., TOKGOZ, S., HAYES, D. & YU, T.H. 2008. Use of US croplands for biofuels increases greenhouse gases through emissions from land-use change. *Science,* **319**, 1238-1240.

SMEETS, E., FAAIJ, A. & LEWANDOWSKI, I. 2005. T*he impact of sustainability criteria on the costs and potentials of bioenergy production. An exploration of the impact of the implementation of sustainability criteria on the costs and potential of bioenergy production, applied for case studies in Brazil and Ukraine*. Netherlands Organisation for Energy and the Environment (NOVEM) B.V., Utrecht, The Netherlands.

SPAROVEK, G., BERNDES, G., EGESKOG, A., MAZZARO DE FREITAS, F.L., GUSTAFSON, S. & HANSSON, J. 2007. Sugarcane ethanol production in Brazil: an expansion model sensitive to socioeconomic and environmental concerns. *Biofuels, Bioproducts and Biorefining,* **1**, 270-282.

STEHFEST, E., ROS, J. & BOUWMAN, L. 2010. *Indirect effects of biofuels: intensification of agricultural production*. Netherlands Environmental Assessment Agency (PBL), Bilthoven, The Netherlands.

STROMBERG, P.M., GASPARATOS, A., LEE, J.S.H., GARCIA-ULLOA, J., KOH, L.P. & TAKEUCHI, K. 2010. *Impacts of Liquid Biofuels on Ecosystem Services and Biodiversity.*

TILMAN, D., SOCOLOW, R., FOLEY, J.A., HILL, J., LARSON, E., LYND, L., PACALA, S., REILLY, J., SEARCHINGER, T., SOMERVILLE, C. & WILLIAMS, R. 2009. Beneficial Biofuels - The Food, Energy, and Environment Trilemma. *Science,* **325**, 270-271.

VAN OORSCHOT, M., ROS, J. & NOTENBOOM, J. 2011. *Evaluation of the indirect effects of biofuel production on biodiversity: assessment across spatial and temporal scales.* Netherlands Environmental Assessment Agency (PBL), 1-8.

ZUURBIER, P. & VAN DE VOOREN, J. 2008. *Sugarcane Ethanol: Contributions to Climate Change Mitigation and the Environment.* Wageningen Academic Publishers, Wageningen, The Netherlands.