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Biofuels and biodiversity: Challenges and opportunities

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ABSTRACT

The use of biofuels can result on a decrease of greenhouse gas (GHG) emissions when compared to fossil fuels. However, the expansion of biofuels crops has been either based on direct or indirect displacement of natural ecosystems or on the use of degraded or marginal lands. The former results in direct habitat loss, whereas the later results in usual agricultural impacts (e.g., soil and biotic contamination and water eutrophication). However, in some circumstances biofuels crops can result on an increase in biodiversity compared to other agricultural crops. Agricultural zoning can mitigate the impacts of land use change (LUC), either direct (dLUC) or indirect (iLUC), whereas the use of wildlife-friendly techniques can mitigate the impacts of agriculture intensification. However, in both cases long-term biodiversity monitoring programs should be established in order to help the decision making process concerning the conflict between the expansion of biofuels crops and the conservation of biodiversity.

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1. General impacts of biofuel crops expansion on biodiversity

The environmental concerns related to the dependence on non-renewable fossil fuels (e.g., air pollution, greenhouse effect, global warming and climate change) have been stimulating the bioenergy production based on agricultural crop biomass in many countries of the Southern and Northern hemispheres (Bhattacharya et al., 2003; Cuvilas et al., 2010; Dauber et al., 2010; Dermibas, 2009; Fernando et al., 2010; Fischer et al., 2010a, 2010b; Goldemberg et al., 2008; Junfeng and Runqing, 2003; Koh and Hoi, 2003; Powlson et al., 2005; Sorda et al., 2010; Williams et al., 2009). Oil palm (*Elaeis guineensis*) in Southeast Asia and recently in Central and South America, soybean (*Glycine max*) and sugarcane (*Saccharum officinarum*) in Brazil, soybean in Argentina, sweet sorghum (*Sorghum vulgare*) in China, maize (*Zea mays*) and soybean in USA, wheat (*Triticum aestivum*), sugar beet (*Beta vulgaris*) and rapeseed (*Brassica napus*) in Northern Europe, and jatropha (*Jatropha curcas*) in Southern Asia and in Africa are the main crops used for liquid biofuels production in terms of cultivated area and volume produced (de Vries et al., 2010; Openshaw, 2000). Other sources of organic matter may be used to produce biofuels, like algae (Ferrell and Sarisky-Reed, 2010), perennial grasses (Tilman et al., 2006a) and woody biomass (for both liquid biofuels and coals) (Bright et al., 2010; Fung et al., 2002; Raison, 2006). However, the expansion of biofuel crops may result in direct or indirect environmental impacts (e.g., alterations in habitat quality, pollution, and bioinvasions), besides conflicts between different sectors of society (Koh and Ghazoul, 2008; Wilcove and Koh, 2010). Such impacts are mostly related to land use change (LUC) and/or agriculture intensification and should be rather considered as of primary not secondary concern as suggested by Milazzo et al. (2013). The evaluation of such impacts should be prioritized by scientists and policy makers in order to establish effective mitigating practices. In addition, long-term biodiversity monitoring programs should be established in biofuels landscapes in order to evaluate their long-term environmental impacts. In the following sections challenges and opportunities concerning the expansion of biofuel crops are presented and discussed.

1.1. Land use change

The expansion of biofuels production has been based on land use change (LUC) directly over pristine ecosystems (dLUC), including biodiversity hotspots (Fig. 1) (Fitzherbert et al., 2008), less

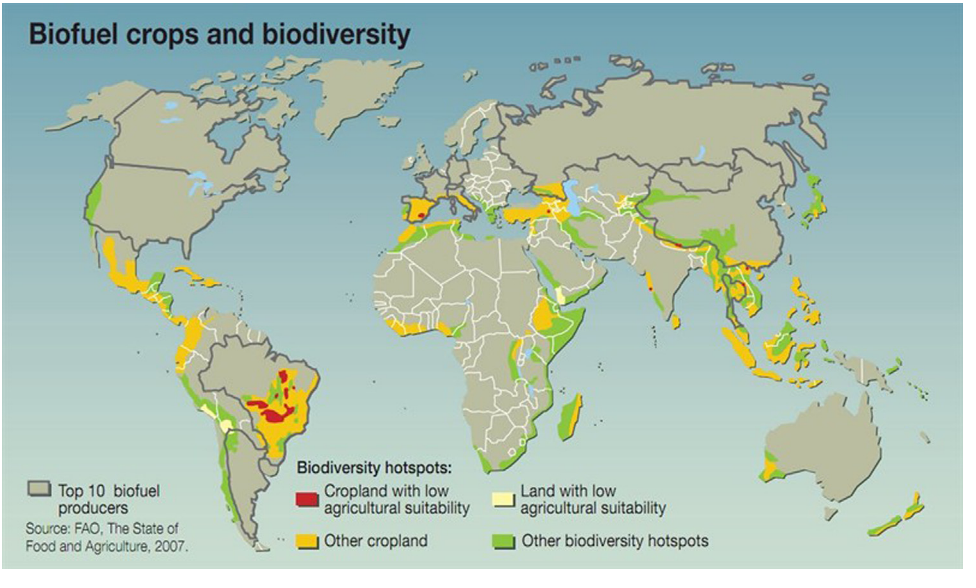

Fig. 1. World distribution of biofuel producers and biodiversity hot spots. (Source: FAO, 2007. The state of food and agriculture).

Table 1
Impacts of biofuels crops on biodiversity (adapted from Joly et al., In press).

Region	Biofuel as landscape matrix	Taxonomic group	Process	Ref.
Argentina	Soybean	Raptors	Decreased diversity	Carrete et al. (2009)
		Caiman	Genotoxicity (in Field-Like Experiment), oxidative stress and DNA damage in wild animals	Poletta et al. (2011, 2014)
		Amphibians	Decrease survival and health	Peltzer et al. (2008)
		Arthropods	Lab. studies reported negative effects in reproduction. Malformations. Spiders wove abnormal web.	Schneider et al. (2009); Benamú et al. (2010)
		Humans	DNA damage, modifications in oxidative balance	Simoniello et al. (2008, 2010)
Brazil	Sugarcane	Rodents	Increased abundance in relation to native vegetation; Spread of emergent infectious diseases (e.g., Hantavirus and Leptospirosis)	Gheler-Costa et al. (2012)
		Rodents	Spread of emergent infectious diseases (e.g., Hantavirus and Leptospirosis)	Verdade et al. (2012)
		Wild canids and felids	Increased abundance in relation to exotic pastures	Dotta and Verdade (2007, 2009)
		Passerine birds	Decreased diversity in relation to degraded exotic pastures	Penteado et al. (2014)
		Birds	Decreased diversity in relation to secondary Atlantic forest	
	Eucalyptus	Birds	Decreased diversity in relation to secondary Atlantic forest	Millan et al (2015); Penteado et al. (2014)
	Soybean	Biodiversity	Reduction of biodiversity in the Cerrado	Janssen and Rutz (2011)
Central America	*Palm oil*	Wildlife	Population reduction by habitat destruction	Höbinger et al. (2012)
USA	Annual crops (i.e., maize and soybean)	Insects (agricultural enemies of food crops)	Decreased abundance in relation to perennial grasslands	Werling et al. (2011)
		Grassland birds	Decreased habitat availability in relation to perennial grasslands	Fletcher et al. (2011); Meehan et al. (2008); Robertson et al. (2010, 2012)
		Migratory birds	Decreased habitat availability in relation to perennial grasslands	Robertson et al. (2013)
	Soybean	Amphibians	Change parasites community	Koprivnikar and Redfern (2012)
Poland	*Populus* sp. plantations	Ground beetles	Species diversity was lower in poplar plantations than adjacent arable farms	Ulrich et al. (2004)
Portugal	*Eucalyptus* plantations	Birds	Lower abundance and diversity than in native habitats	Pina (1989)
		Macroinvertebrates	Lower abundance and diversity in streams that cross *Eucalyptus* plantations than those that run through native land covers	Abelho and Graça (1996)
Spain	*Eucalyptus* plantations	Birds	Lower abundance and diversity than in native habitats	Calviño-Cancela (2013)
Sweden	*Salix* plantations	Birds	Taller plantations supporting higher diversity of species; Diversity was also higher in planted *Salix* sp. forest than in	Berg (2002)

Table 1 (*continued*)

Region	Biofuel as landscape matrix	Taxonomic group	Process	Ref.
			other open farmland sites dominated by other crop-fields	
UK	*Miscanthus*	Flora and birds	Decreased diversity in relation to short rotation coppice (SRC) willow or poplar	Rowe et al. (2009)
		Flora, small mammals and birds	Higher abundance in ecotone areas	Semere and Slater (2007)
		Birds	Higher abundance and diversity of birds when compared with wheat crops	Bellamy et al. (2009)
	Salix plantations	Flora	Plant richness and diversity was higher in short rotation willow coppice than in grassland	Fry and Slater (2008)
SE Asia	Palm oil	Vertebrate species	Decreased diversity	Danielsen et al. (2009)
		Forest birds	Decreased diversity	Sodhi et al. (2005)
		Insectivorous birds	Predation on herbivorous insects that attack palm oil plants	Koh (2007)
		Orangutan	Population reduction	Meijaard et al. (2012); van Schaik (2004)
		Birds, ants, beetles	Diversity reduction. Loss of species of big size. Increase abundance in Palm Oil plantations of rare (or absent) species in the original forest.	Senior et al. (2013)
		Crocodiles	Population reduction of endangered and critically endangered species due to habitat loss.	Stuebing et al. (2006); Staniewicz and Behler (2011), Bezuijen et al. (2014)
		Mammals	Change in community structure, density reduction of deer and increasing of wild boar populations.	Luskin et al. (2014)
		Reptiles, avian, and mammals	Killing of wildlife, population reduction.	Azhar et al. (2013)
		Birds and butterfly	The conversion of primary or secondary forests to oil palm results in significant biodiversity losses; conversion of rubber cropland to oil palm has a lower impact.	Koh and Wilcove (2008)
		Sumatran elephant (*E. maximus sumatranus*)	Population declined 84% over the last 23 years, due to an increase of human-elephant conflict associated with the replacement of forest by palm oil plantations.	Uryu et al. (2008)
		Sumatran tiger (*Pantheratigrissumatrae*)	Population declined 70% over the last 23 years, due to habitat fragmentation and decrease in prey availability associated with palm oil plantations implementation.	Uryu et al. (2008)
	Palm Oil and rubber plantations	Birds	The conversion of lowland forest to plantations reduced 60% of the species richness of at least 60%, mainly insectivores and frugivores birds.	Aratrakorn et al. (2006)
India	*Eucalyptus* plantations	Asian elephant (*Elephas maximus*)	*Eucalyptus* plantations are more used than tea and coffee plantations.	Kumar et al. (2010)
Africa	Palm oil	Primates	Diversity reduction	Linder (2013)

profitable (usually food) crop lands (Scharlemann and Laurance, 2008), and degraded lands (Plieninger and Gaertner, 2011). Indirect displacements of pristine ecosystem as a consequence of secondary dLUC by crops formerly displaced by biofuel crops – or simply indirect land use change (iLUC) – can also occur as a result of biofuel crops expansion (Lapola et al., 2010). dLUC and iLUC result in biodiversity loss due to the direct loss of wildlife habitat (Bindraban et al., 2009; Borjesson and Tufvesson, 2011; Borzoni, 2011; Fargione et al., 2008, 2010; Fischer et al., 2010b; Fritsche et al., 2010; Gawel and Ludwig, 2011; George et al., 2012; Goldemberg et al., 2008; Gutiérrez-Velez et al., 2011; Janssen and Rutz, 2011; Kocoloski et al., 2009; Koh, 2007; Koh and Wilcove, 2009; Koh et al., 2011; Leal et al., 2013; Overmars et al., 2011; Pedroli et al., 2013; Phalan, 2009; Phalan et al., 2013; Richards et al., 2012; Van Stappen et al., 2011). The use of degraded land for biofuels production is usually associated with an intensification of agricultural practices (Fernando et al., 2010; Prins et al., 2011), including an increase in the use of agrochemicals with the consequent contamination of the biota and the physical environment (Ceotto, 2008; Hellmann and Verburg, 2010; Meche et al., 2009; Schiesari and Grillitsch, 2011; Verdade et al., 2012).

Besides the usual impacts caused by agriculture (Robertson et al., 2011), the expansion of biofuels production can affect food security and cause pollution, invasion of exotic species, the spread of wildlife-related diseases and alterations in communities structures and biodiversity. However, in some circumstances, biofuel crops may have a positive impact on biodiversity in relation to other agricultural land uses (Milder et al., 2008) (Table 1).

Perennial grasslands used for biomass production may enhance avian diversity (i.e., species richness and abundance), including migratory species, in relation to corn fields in USA (Fletcher et al., 2011; Robertson et al., 2010, 2012, 2013). The higher habitat quality of perennial grassland is due to a combination of an increased vegetation density and food (i.e., arthropod) availability (Robertson et al., 2012). In addition, perennial grasslands used for bioenergy production could conserve arthropod species who are natural enemies of food crops plagues, usually less abundant in annual crops like corn (Werling et al., 2011). However, in UK birds and plants are less diverse in areas dominated by *Miscanthus*, an exotic grass originated in Southeast Asia used for biomass production, than on short rotation coppice (SRC), willow or poplar (Rowe et al., 2009). Semere and Slater (2007) also detected that ground flora, small mammals and most birds were more abundant in ecotone areas (i.e. crop boundary areas) than within *Miscanthus* fields, suggesting the importance of maintaining spatially heterogeneous agricultural landscapes. However, Bellamy et al. (2009) showed that in UK, when compared with other field crops (e.g. wheat), *Miscanthus* fields supported a higher abundance and diversity of birds, probably due to better shelter provided by this crop. Although *Miscanthus* is being grown commercially mostly in northern European Union (EU), mainly in the UK, for direct use in power stations (Heaton et al., 2004), several trials have been implemented throughout Europe and Eastern Asia (Lewandowski et al., 2000), but no large scale plantations have yet been implemented. However, in some eastern European and western Asia countries, such as Lithuania, Latvia, Romania, Georgia, Belarus and Azerbaijan the potential for energy production from this perennial grass is high, which together with poplars and willows plantations may provide one third of those countries commercial energy demands (Fischer et al., 2005).

Sugarcane plantations for ethanol and sugar production in Brazil might expand from 8 Mha and to 14 Mha by 2016 (UNICA, 2008). Such expansion is predominantly occurring over degraded exotic pastures in Southeastern Brazil; however, the iLUC is considered a relevant driver to deforestation on the Amazon basin (Lapola et al., 2010; Martinelli and Filoso, 2008). Although direct deforestation due to logging and subsequent livestock production in the Amazon appears to be independent from the sugarcane expansion in Southeastern Brazil, water eutrophication and soil pollution might be locally relevant (Verdade et al., 2012). In addition, passerine birds are less diverse on sugarcane fields than on degraded exotic pastures likely due to their lack of vegetation structure and food resources (Penteado et al., 2014). However, there is an increase in rodents abundance associated with the expansion of sugarcane fields (Gheler-Costa et al., 2012), including the capybaras (*Hydrochoerus hydrochaerus*), the largest living rodent (Ferraz et al., 2009; Verdade and Ferraz, 2006). These rodents support a relatively large number of predators, including wild canids and felids (Dotta and Verdade, 2007, 2009; Verdade et al., 2011) and raptors (Penteado et al., 2014). On the other hand, small rodents from sugarcane fields have been associated with the emergence of wildlife related diseases like Hantavirus and Leptospirosis (Verdade et al., 2012) and capybaras have been associated with the spread of mites (*Amblyomma cooperi*) that host *Rickettsia rickettsii*, the pathogen that causes the Brazilian spotted fever (Labruna, 2012). The expansion of biofuels may be directly or indirectly related to the emergence of other infectious diseases (Patz et al., 2008).

In Brazil, although the total area covered by soybean field crops are twice as big as the area covered by sugarcane, besides the biodiversity loss caused by LUC, the impact of soybean production for both food and biofuels are still mostly unknown (Janssen and Rutz, 2011; Verdade et al., 2012). However, in Argentina raptors decreased in response to the expansion of soybean production (Carrete et al., 2009) and caiman living in wetlands nearby soybean plantations presented high levels of DNA damage and oxidative stress possibly due to residues of gliphosate (Poletta et al., 2011, 2014). Similar problems have also been described for amphibians (Koprivnikar and Redfern, 2012; Peltzer et al., 2008), arthropods (Benamú et al., 2010; Schneider et al., 2009), and even humans (Simoniello et al., 2008, 2010).

Eucalyptus plantations in Southeastern Brazil, either for paper or coal production, are used by endangered wildlife (e.g., giant anteater, *Myrmecophaga tridactyla*, and South American tapir *Tapirus terrestris*). These species apparently use the *Eucalyptus* stands as habitat, not only as a permeable matrix (Dotta and Verdade, 2011; Timo et al., 2014). In addition, a considerable diversity of plants, including endangered species like *Araucaria angustifolia* and *Dalbergia nigra*, can be found in *Eucalyptus* plantations, as scattered trees or in the understory vegetation (Athayde et al. 2014; Gabriel et al., 2013). Although birds' α-diversity is significantly smaller inside *Eucalyptus* plantations than in the surrounding secondary native vegetation, the presence of native trees, stimulated by the certification process, enhances birds' diversity inside *Eucalyptus* plantations (Millan et al. 2015). A similar pattern was detected in Eurasia, where *Eucalyptus* plantations cover approximately 12 Mha (Iglesias-Trabado and Wilstermann, 2008). In Southwestern Europe, studies have showed that *Eucalyptus* plantations (mainly devoted to paper pulp production, although bioenergy production importance is increasing; DNFF, 2010) have direct and indirect effects upon wildlife: while supporting lower bird's abundance and diversity than native ecosystems (Calviño-Cancela, 2013; Pina, 1989), *Eucalyptus* plantations affect macroinvertebrate communities of streams that cross plantations, decreasing its abundance and diversity due to alterations on the physic-chemical characteristics of streams, associated for example with leaf and bark litter fall and decomposition differences (Abelho and Graça, 1996). However, in areas where native vegetation was fragmented by agroforestry systems, *Eucalyptus* plantation may assume an important ecological role. For example, in Anamalai Hills (Western Ghats, India), an area dominated by tea, coffee, and *Eucalyptus* plantations, interspersed by rainforest fragments, *Eucalyptus* stands are important habitats for the Asian elephant (*Elephas maximus*), by providing cover, food and shelter, mitigating human-elephant conflicts (Kumar et al., 2010).

Oil palm crops currently covers 13.5 Mha in Southeast Asia, mostly (80%) in Indonesia and Malaysia (Fitzherbert et al., 2008). More than 50% of the recent (1990–2005) palm oil expansion is directly related to dLUC (i.e., deforestation) (Koh and Wilcove, 2008; Sodhi et al., 2010a, 2010b). The annual deforestation rate in Malaysia has been over 22,000 ha per year since the 1980s (Koh and Hoi, 2003). Converting forest into palm oil crop is more profitable than preserving it for carbon credits traded in compliance markets (Butler et al., 2009). Such tendency can be supported by the international market (Lenzen et al., 2012) and might result on a massive biodiversity loss (Sodhi et al., 2004) in specialist forest birds (Sodhi et al., 2005). Palm oil plantations hold less than half (38%) of the vertebrate species than the primary forest with only 23% found in both environments (Danielsen et al., 2009). However, insectivorous birds prey on insects that attack palm oil plants (Koh, 2008a). Including all taxa, only 15% of all the species detected in primary forests where also recorded within palm plantations (Fitzherbert et al., 2008). Oil palm plantations have modified community structure (Azhar et al., 2013; Luskin et al., 2014; Senior et al., 2013), with increasing population of few species, but possibly reducing many others (Senior et al., 2013), like crocodilians (Bezuijen et al., 2014; Staniewicz and Behler, 2011; Stuebing et al., 2006), orangutans (Meijaard et al., 2012; van Schaik, 2004) and other primates (Linder, 2013). This selective reduction of large-sized species may reduce proteins income to the local human population, increasing hunting pressure over other species (Luskin et al., 2014). The expansion of palm oil in Latin America is expected to produce similar impacts to wildlife (Höbinger et al., 2012).

J. curcas is a native plant of the Neotropics that has been used as fuel substitute during the World War II in some countries, but currently used as an energy source in Africa (e.g., Uganda, Mozambique, Zambia) and Asia (e.g., Thailand, China and India) (Kumar and Sharma, 2008; Li et al., 2007). Although *Jatropha* plantations reached only 900,000 ha by 2008, in Asia (85%), Africa (13%) and Latin America

(2%), this area is expected to grow exponentially, especially in China, reaching 12.8 Mha in 2015 (Contran et al., 2013; Froger et al., 2010; Phalan, 2009). In India, the national plan for *Jatropha* production expects to convert 3 Mha of forests considered "understocked". Future studies should prioritize the impacts of such LUC on the environment and biodiversity.

Willow and poplar plantations cover 9.2 Mha including China (8,037,600 ha), France (236,000 ha), Iran (160,000 ha), Turkey (125,000 ha), Spain (105,000 ha), Italy (121,430 ha), Argentina (56,400 ha), Romania (19,505 ha), and Sweden (11,100 ha) (FAO, 2012; Rosenqvist et al., 2000). Several studies have assessed the impacts of these wooded plantations on biodiversity, mainly in northern Europe. The plantation age, and consequently its structure, influences the bird community, with taller plantations supporting higher species diversity, than on open farmland sites dominated by other crop-fields (Berg, 2002). This positive effect of *Salix* plantations on biodiversity when compared to other agroecosystems was also described in Wales, where plant richness and diversity within short rotation willow coppice was greater than in grassland (Fry and Slater, 2008). For poplar plantation a distinct pattern was detect in Northern Poland, where ground beetles' species diversity was lower than on adjacent arable land, suggesting that poplars plantations did not enhance overall regional diversity (Ulrich et al., 2004).

1.2. Agriculture intensification

Pollution from agrochemicals (e.g., fertilizers and pesticides) associated with biofuels production may cause a relevant impact on terrestrial and aquatic organisms. The expansion of biofuels production in Brazil—in special sugarcane on the Southeast and soybean on the Amazon basin—is resulting on an increase on the use of compounds that have been suspended by international conventions, including those of priority concern for exhibiting environmental persistence and/or having the potential to elicit neurotoxic, reprotoxic, carcinogenic, or endocrine-disrupting effects in humans and wildlife (Schiesari and Grillitsch, 2011). In addition, the annual exposure of bare soil rich in aluminum can result in contamination of freshwater fish (Meche et al., 2009). However, the fertilizer and pesticides inputs in biofuel production vary with the type of species used, with corn and sugarcane reaching the higher inputs and switchgrass (*Panicum* sp.) and microalgae the lower (Groom et al., 2008).

Woody biomass-to-liquid production (BTL) may locally increase eutrophication and acidification not easily detected (Sunde et al., 2011). Air pollution can also be a concern in biofuels production (Williams et al., 2009) including anthropogenic emissions of NH_3 (Erisman et al., 2007). Indirectly, wildlife may also be globally affected by water limitations associated with water requirements of biofuel production and agriculture intensification. This can vary greatly with the type of production and region. For example, the water involved for producing the same amount of energy from corn (for ethanol production) irrigation is a fourth of that used in soybean (for biodiesel production). In addition, 30% more water is required to produce the same energy from sorghum (for ethanol production) in Texas than in Nebraska (Dominguez-Faus et al., 2009).

Biofuel production is also likely to increase the risks and costs associated with invasive species. These risks can occur as a direct consequence of the species and genotypes used to produce biofuels, but also as a consequence of the invasion of other taxa somehow fostered by the habitat changes associated with biofuel production (Sala et al., 2009). Invasive species have driven native species to extinction, altered the composition of ecological communities, changed patterns of periodical events and altered ecosystem processes (Vitousek et al., 1987). This can be a relevant problem in Africa (Blanchard et al., 2011; Witt, 2010) and Europe (Genovesi, 2010), where biofuels production has been based on the introduction of exotic species.

2. Mitigating impacts of biofuels production on biodiversity and the environment

The existing mitigating measures for the environmental impacts caused by dLUC and iLUC are basically distinct but complementary. The former is usually—although not always effectively—controlled by agriculture zoning in which limits for the expansion of biofuel crops over pristine ecosystems are established by each country. The later is predominantly focused on the use of wildlife friendly agricultural practices. Both approaches are complementary in terms of public policy (Charles et al., 2007; Lovett et al., 2011; Soderberg and Eckberg, 2013) and market (Di Lucia, 2010; Palmujoki, 2009). However, in order to be effective both strategies depend on a global network of long-term monitoring program as discussed on the following sections.

2.1. Agriculture zoning

Agriculture zoning for biodiesel production should be based on edaphic and hydrological limitations for agriculture (Lal, 2008), but also on environmental sustainability which can be attained by banning of dLUC and use of degraded land (Groom et al., 2008; Joly et al., 2010), as follows.

2.1.1. Banning or drastic restriction of dLUC

Financial incentives to reduce carbon emission from deforestation and forest degradation (REDD) have been proposed as economic compensation for landowners to reduce deforestation (Butler et al., 2009; Kileen et al., 2011; Visseren-Hamakers et al., 2012). However, this strategy is market dependent (Bateman et al., 2010; Koh and Wilcove, 2007; Persson, 2012), what makes risks unpredictable even for economists (Taleb, 2007).

2.1.2. Use of degraded land

The expansion of biofuel crops over degraded lands instead of over pristine ecosystems and food crop lands has obvious advantages for sustainability and food security (Fitzherbert et al., 2008; Henneberg et al., 2009; Koh and Ghazoul, 2010; Obidzinski et al., 2012; Plieninger and Gaertner, 2011; Ravindranath et al., 2011; Stoms et al., 2012; van Vuurven et al., 2009). However, degraded lands are usually associated with edaphic and hydrological limitations, which require studies to develop plant species adapted for such circumstances (Li et al., 2010). In addition, an appropriate agriculture zoning should be complemented by wildlife-friendly agricultural practices, as discussed below.

2.2. Wildlife friendly agricultural practices

The environmental impacts of agriculture can be mitigated by either improving productivity in smaller areas or reducing productivity in larger areas (Green et al., 2005). When it works the former results in concentrated highly productive crop fields and more pristine areas maintained for conservation (Buckeridge et al., 2012; Koh et al., 2009). The later results in agricultural landscapes composed by an agroecosystem (i.e., crop fields) as landscape matrix and conservation areas, including waste land and usually secondary remnants of native vegetation with some conservation value (Koh, 2008b; Metzger et al., 2010; Ranghanatan et al., 2008; Smith et al., 2008; Smith and Gross, 2007; Verdade et al., 2014a; Wiens et al., 2011).

The conservation of remnants of native vegetation in agricultural landscapes can increase not only the matrix permeability for specialist species but also the habitat quality of the matrix *per se*, thus increasing the landscape β-diversity (Verdade et al., 2014a). As a result, there must be a local improvement of ecosystem services (Berry and Paterson, 2009; Gasparatos et al., 2011; George et al., 2012). Such strategy implies in the assumption of multifunctionality of agricultural landscapes (Martinelli et al., 2010) including the production of domesticated species and the conservation of wild species (Verdade et al., 2014a).

As an example of the influence of agricultural practices, the suspension of pre-harvest fire in sugarcane fields in Southeastern Brazil result in a likely reduction in the mortality of mesopredators with a consequent decrease in the abundance of small rodents (Gheler-Costa et al., 2013). The impact of such process on the spread of emergent infectious diseases (e.g., Hantavirus) should be prioritized by future studies. In addition, the maintenance of sparse trees and some understory vegetation in *Eucalyptus* plantations, also in Southeastern Brazil, can result on an increase of birds' α-diversity (Millan et al. 2015)". However, the age of plantation can affect the frequency of occurrence of medium to large mammals (Timo et al., 2014) and the direction of plantation lines can affect the dispersal of small marsupials (Prevedello and Vieira, 2011).

Table 2
Number of studies assessing fauna and flora biodiversity in biofuels crops in Europe and USA (from Dauber et al., 2010).

Taxonomic group	Willow	Poplar	Willow+Poplar	Sweetgum	Perennial grasses (*Miscanthus* sp.)	Reed canary grass (*Phalaris* sp.)	Switchgrass (*Panicum* sp.)
Birds	7	3	5		4	2	1
Mammals	1	2	1		3	2	
Butterflies	3				2	1	
Ground arthropods		9	5	1	5	3	1
Canopy invertebrates	2	1			2	1	
Earthworms	3	2	1		2		
Other soil fauna	1	1					
Plants	5	4	2		2	2	

The examples above are spatially restricted to provide a broad view of the influences of agricultural practices on patterns of biodiversity. Multidisciplinary agronomic, socioeconomic and ecological research is required to develop wildlife-friendly agricultural practices in other biofuel crops (Henneberg et al., 2009; Popp et al., 2011; Ridley et al., 2010).

2.3. Biodiversity and environmental monitoring in agricultural landscapes

The majority of studies assessing the impacts of bio-energy crops on biodiversity is regionally and taxonomically biased and temporarily restricted (seasonal or yearly limited). For example, Dauber et al. (2010) reviewed 47 publications from western countries (USA and Europe) which evaluated the impact of some biomass crop productions on flora and fauna (Table 2). Their study revealed that birds and ground arthropods were the most represented groups in the reviewed studies, which were mostly focused on willow, poplar and perennial grasses plantations. This limited view of the overall impact of such productions demonstrates that the assessment of the long-term impacts of biofuels production on biodiversity requires a global network of long-term monitoring program (Sodhi et al., 2010a; Tilman et al., 2006b). Such program should include life cycle impact assessments (LCA) of biofuel crops (Bare, 2011; Finnveden et al., 2009; Markevicius et al., 2010; Pennington et al., 2004; Rebitzer et al., 2004; Reinherdt and von Falkenstein, 2011; von Blottnitz and Curran, 2007; Weiss et al., 2012), associated with the use of geographic information systems (GIS) (Geyer et al., 2010) and validated simulation models (Bastolla et al., 2005; Koh and Ghazoul, 2009; Magnusson et al., 2014; Zhang et al., 2010). Sampling procedures should be systematized including methodological adaptations and technological innovations in order to reduce methodological uncertainties (e.g., Gao et al., 2011; Lyra-Jorge et al., 2014; Penteado et al., 2014). The database generated by the distinct sampling sites of such global network should be interoperable in order to connect patterns of diversity with complexity of processes in distinct biomes and landscapes (Verdade et al., 2014b). By this approach it would be possible to generate effective indicators of the former (e.g., Scharlemann, 2008) based on the monitoring of the later (e.g., Wilbur, 1997).

The expansion of biofuel crops, together with other bioenergy sources, might have a global positive impact based on the reduction of greenhouse gasses (GHG) emissions. However, regional and local impacts on the environment and biodiversity may be relevant. Such impacts should be properly identified and monitored in order to be mitigated. In such scenario there is a great demand for multidisciplinary research and capacity building both locally and globally.

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