The impacts of biofuel production on biodiversity:
A review of the current literature

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1 Abstract

The production of liquid biofuels is rapidly increasing. Governments are setting targets to increase the proportion of biofuels in their energy mix for the purposes of climate change mitigation, energy security and rural development. Most research to date has focused on the performance of biofuels in reducing carbon emissions, with questions being raised over their potential in this respect. Far less attention has been paid to the potential impacts of biofuels on biodiversity.

The biodiversity impact of biofuels will depend on the biofuel crop and the previous land use. Biofuels can be beneficial to biodiversity when appropriate crops are grown in suitable areas. Furthermore, if they contribute to climate change mitigation, they have the potential to be indirectly beneficial to biodiversity as a whole. However, biofuels have already been shown to negatively impact biodiversity when direct conversion of natural ecosystems or indirect land conversion of non-degraded land occurs. The expansion of biofuel production in the tropics has resulted in the loss of tropical forest and wetlands, and in temperate regions biofuel production has encroached into set-aside lands. Biofuel feedstock plantations (particularly oil palm and maize plantations), have been shown to support far lower levels if biodiversity than natural ecosystems, and can cause soil erosion and the pollution of watercourses. How a feedstock plantation is managed influences the level of biodiversity impacts. Well managed plantations can in some instances prove beneficial to biodiversity especially if these are on degraded or marginal lands.

As the demand for biofuels increases in the future so will the land requirements. Conversion of natural land is likely to be detrimental to biodiversity. Furthermore post-conversion management can also negatively impact biodiversity through the pollution of fertilizers for example. Some maintain that the 'next generation' of biofuels will require less land or be more productive and therefore reduce negative biodiversity impacts though there is much uncertainty surrounding this. The introduction of sustainability standards is one option to reduce the biodiversity impacts of biofuel production. However, the development and implementation of these standards is proving difficult, largely due to the lack of accepted definitions for key terms such as 'high biodiversity' and 'degraded' lands. In any case, it is likely that sustainability standards will only be part of the solution, and will need to be combined with improved land use planning.

2 Introduction

Bioenergy accounts for approximately 10% of global energy production, and can be defined as energy produced from any source of biomass; i.e. plants, animals, and organic waste. The use of bioenergy ranges from traditional energy in rural populations to the use of liquid biofuels in the transport sector. This paper focuses specifically on the production of 'biofuels', narrowly defined in this case as liquid biofuels for transport.

Although biofuels can in principle be produced from any organic source (often termed "feedstock"), most of the current or 'first generation' biofuels are based on food crops. 98% of current biofuel production involves the production of ethanol from sugars and biodiesel from oil seeds (Msangi *et al.* 2008). The main crops used in ethanol production are sugar cane and maize, with oil palm and rapeseed most often used to produce biodiesel (Table 1). Different feedstocks are more or less efficient in the production of bioenergy (Sheil *et al.* 2009) and some feedstocks also provide useful co-products such as oilcake as animal feed. In discussions on the first generation biofuels, the type of feedstock used is pivotal because of the wide variety used (Table 1).

'Second generation' biofuels, where advanced technology is used to break down lignin and cellulose to convert biomass and waste products into fuel, are currently in development; as are 'third generation' biofuels produced from algae. The exact criteria that result in the labelling of 'first', 'second', and 'third' generation biofuels are not clearly defined. In this report, 'first generation' biofuels will be taken to mean any biofuel currently in large-scale production.

Table 1. Major biofuel sources. Adapted from Biemans *et al.* (2008)

1st generation		2nd generation	3rd generation
Biodiesel	**Bioethanol**		
Palm oil Rape seed Sunflowers Soy beans Jatropha	Corn Sugar cane Sugar beets Wheat	Willows Poplars Grass Agricultural waste products Forestry waste products	Algae

Biofuels account for only a small proportion of energy use, currently providing less than 1% of the global energy supply (FAO 2008b) and approximately 1.8% of the global liquid transport fuel use (Howarth *et al.* 2009). Despite its small current contribution to transport energy, the production of liquid biofuels has increased rapidly in recent years (Gallagher 2008; GBEP 2008). This increase can be attributed in part to government targets and subsidies (such as the EU's renewable energy target), which have been established to promote the use of biofuels for reasons of energy security, climate change mitigation, and rural development (Johnston *et al.* 2009).

This rapid increase has led to much debate over the social and environmental impacts of biofuel production. In particular, recent literature has focused on whether and how biofuels can achieve an absolute reduction in carbon emissions where feedstock plantations replace carbon-rich natural ecosystems. In addition to the concerns surrounding the mitigation potential of certain feedstocks, the potential unintended negative impacts on biodiversity are being considered (Danielsen et al. 2009; Fitzherbert et al. 2008). In particular, the impacts of indirect land use change (iLUC), where the use of agricultural land for biofuel production results in the displacement of agriculture into natural ecosystems, have also been discussed (Gallagher 2008).

This report reviews the current state of knowledge[1] on the biodiversity impacts (both positive and negative) of biofuel production, with an emphasis on the potential influence of current and future government policies. Although the focus is primarily on first generation biofuels, second and third generation biofuels are also discussed. Finally, the potential for sustainability criteria to ameliorate biodiversity impacts is assessed.

3 Background

3.1 The influence of biofuel policy targets

A number of countries have integrated biofuel targets into their renewable energy policies in recent years. The EU, the USA, Canada, Brazil, Argentina, Colombia, China, New Zealand and Japan all have targets in place for the use of biofuel in transport; and the list is growing Steenblik 2007). The EU, for example, has set a 10% target for biofuel use within the transport sector by 2020, and in the UK, biofuel already accounts for 2.5% of the fuel blend. To help meet such targets, biofuel production is heavily subsidised in some countries (Doornbosch & Steenblik 2007). Financial support measures from the USA, Europe, Switzerland, Australia and

[1] This literature review is based on the literature published in the English language. It is acknowledged that this may be a limitation to this review since it may fail to incorporate studies and findings that have been undertaken in some of the major feedstock growing countries.

Canada reportedly totalled $11 billion in 2006; the majority for biodiesel in Europe, and ethanol in the USA (Eickhout *et al.* 2008).

It is generally agreed that these targets and financial incentives are behind the current and likely future increase in biofuel production (Ravindranath *et al.* 2009). The influence of such policies can be underlined through modelling of demand under different policy measures. It has been suggested, for example, that ethanol production would be reduced by 30% and biodiesel by more than 50% without policy measures (FAPRI 2008); with OECD projections similarly suggesting that the removal of biofuel policies and subsidies would reduce ethanol production in the US by 20%, and by 80% in Canada and Europe (Searchinger 2009).

In some countries that do not currently have targets, subsidies or policies for biofuel production for domestic use, production is driven by the export market. This is particularly the case in Indonesia, where 18 million hectares (Mha) of land is already used for palm oil production, and further large-scale plantations are planned. However, several African countries are beginning to explore the potential of domestic biofuel targets, as are a number of palm oil producers (Dufey 2006). If such policies were to be developed, this would further increase demand for biofuel production. Thailand, for example, is developing targets equivalent to 10% blend of biodiesel by 2012 (AEA Technology 2008).

The Convention on Biological Diversity (CBD) has emphasised the need for the adoption of adequate policy frameworks to ensure that the production of biofuels is sustainable. Parties are urged to promote sustainable production and use, taking into account the full life cycle and acting in accordance with the precautionary principle (UNEP/CBD/COP/DEC/IX/2). However, current targets appear to have been developed with little consideration for the environmental consequences of biofuel production (Doornbosch & Steenblik 2007).

3.2 Global projections of land requirements and availability

Currently, 13.8 Mha of land is used for biofuel production across the USA, the EU, Brazil and China, less than 1% of the 1500 Mha currently used as cropland (Gallagher 2008). However, the land required for increased biofuel production will be in addition to the agricultural demand (OECD/FAO 2008). The US and Global Agricultural Outlook (FAPRI 2008) projects large increases in global coarse grain area due to increased demand, in addition to a 14% increase in the harvested area of sugarcane and a 35% increase in oil palm area by 2017/18 due to targets set by the EU and the US, and the likelihood of increased biofuel targets in Brazil, China, Argentina and India.

Different scenarios and models have led to varying projections of the land area required for demand-driven biofuel expansion. These estimates range from 56 – 2,500 Mha, where the lower bound (Gallagher 2008) takes into account land savings through the production of co-products, and the upper bound (Gurgel *et al.* 2008) is an estimate for all biomass fuel requirements under a policy to severely limit the usage of fossil fuels. To put this in context, an estimate towards the middle of this range (850 Mha) is equivalent to half of the current global crop land (Muller *et al.* 2008). Estimates vary because the amount of land required will depend upon the biofuel crops modelled[2] as well as assumptions on efficiency, co-products, and land productivity (Howarth *et al.* 2009). For example, a recent study by Ravindranath *et al.* (2009) estimates that 118 – 508 Mha of land would be required to meet a target of 10% biofuel in transport fuels globally, depending on the main crops used to meet targets.

The availability of land for biofuel production is another question entirely. Estimates vary depending on the land use data and whether definitions of 'available' land exclude forest, cropland, etc. Optimistic estimates suggest that there could be up to 1,215 Mha of land available, whereas pessimistic estimates can be as low as 400 Mha (Gallagher 2008). Although it is difficult to say with any certainty, the ranges presented in the literature suggest a potential deficit between land availability and projected land requirements to meet biofuel production targets. The pessimistic scenario presented in the Gallagher review projects a land deficit of approximately 200 Mha when additional food and feed requirements are taken into account (Gallagher 2008). No estimates in the literature reviewed here took land requirements for other climate mitigation policies such as afforestation and wind energy into consideration, or factored in competition for water resources.

3.3 Biofuels and climate change mitigation

Biofuel targets have largely been set as part of renewable energy policies in the context of climate change mitigation. Biofuels can undoubtedly contribute to climate change mitigation when grown in appropriate areas. For example, recent studies have suggested that when sugarcane is used to produce ethanol, 80-100% greenhouse gas savings could be achieved, and that oilseed rape production for biodiesel can similarly achieve emissions savings of 20-85% (Howarth *et al.* 2009). Where biofuels achieve real emissions reductions, this would have biodiversity benefits through reducing climate change impacts (UN-Energy 2007). This is an important trade-off to keep in mind when considering some of the potential negative impacts on biodiversity resulting from the cultivation of feedstocks. Moreover, the time frame, over which the impacts on biodiversity resulting from biofuel production are examined, also needs to be considered. Indeed, Eickhout *et al.* (2008) suggest that in the short to medium

[2] Different biofuel crops such as sugar cane, oil palm and maize have vastly different land requirements. Oil palm plantations, for example, require less land than alternative biofuel crops to produce comparative amounts of fuel (Sheil *et al.* 2009)

term, where biofuel production replaces natural ecosystems, including those with lower carbon storage values, the negative effects of land use change on biodiversity are likely to outweigh any benefits that may be gained from climate change mitigation. Biofuels may further aid climate change mitigation with the development of more efficient biomass-use technologies such generating energy through the combustion of bagasse (Machado-Filho 2008).

However, recent research has suggested that the production of energy crops may do little to mitigate climate change where they replace natural ecosystems; even increasing emissions by as much as 17-420 times compared to that of fossil fuels (Fargione *et al.* 2008; Righelato & Spracklen 2007; Searchinger *et al.* 2008; Gallagher 2008). One recent study has estimated that land conversion for biofuels could result in emissions of 753-1825 Mt CO_2 per year, compared to the 840 Mt CO_2 that is emitted from 10% petrol consumption (Howarth *et al.* 2009; Ravindranath *et al.* 2009). Furthermore, although the use of palm oil can achieve large greenhouse gas emissions savings, the conversion of rainforest and peat soils can actually result in 800-2000% higher emissions than equivalent fossil fuels (Howarth *et al.* 2009). However, some question the assumptions made in these studies, and suggest that an improved understanding of the drivers of land use change would lead to more positive conclusions about the potential for biofuels in climate change mitigation (Kline & Dale 2008). Moreover, the type of feedstock analysed in these studies, as well as where and what land use type is replaced by the feedstock, contributes to the differences between studies.

4 Impacts on biodiversity

The impacts of biofuels on biodiversity will depend greatly on the type of crops that are planted and the previous land use. As habitat loss and degradation are major threats to biodiversity (Hennenberg *et al.* 2009), direct conversion of natural ecosystems and indirect land use change to accommodate biofuel production is likely to be detrimental to biodiversity (GBEP 2008). Indeed negative impacts due to such changes have been documented (Royal Society 2008; Gallagher 2008). However, plantations on marginal or degraded lands could have positive effects on biodiversity (Eickhout *et al.* 2008). The off-farm impacts of biofuel feedstock plantations have been less well documented but can include, depending on the management regime, reduced water availability, soil erosion, and the spread of invasive species. It is difficult to generalise the impact of 'biofuels', as each biofuel crop has its own set of advantages and costs, although some authors suggest that some have higher aggregated environmental costs than fossil fuels (Zah *et al.* 2007). In general terms, biodiversity loss occurs when high biodiversity land is converted into plantations that contain lower levels of biodiversity. The impacts on biodiversity are therefore a function of the biodiversity present prior to land conversion, the biodiversity present after the land has been converted for biofuel feedstock production, and the 'off-farm' impacts of the biofuel feedstock plantations on the surrounding areas.

4.1 Loss of natural land

4.1.1 Direct conversion of natural and semi-natural ecosystems

Much of the discussion on the biodiversity impacts of biofuel production surrounds the direct impacts of land conversion, where natural ecosystems are replaced by biofuel feedstock plantations.

There is already considerable land use pressure from agriculture worldwide, and this is projected to increase over the coming decades (FAO 2008b). The land requirements for biofuel production at the scale discussed in Section 3 will be a large additional demand. One study has estimated that 0.4 Mha – 114 Mha of natural land could be lost due to biodiesel production alone, depending upon the feedstock used for production and whether current agricultural land was used (Koh 2007). It should be noted that biodiversity loss will not be proportional to area loss, as some areas are much more diverse than others (Sala *et al.* 2009). Tropical forests, which are particularly vulnerable to conversion to biofuel plantations, contain a high proportion of global biodiversity (Sala *et al.* 2009). On the other hand, loss of natural land may not occur, or not occur to the extent estimated, if agricultural production is optimized and yields increased on existing agricultural land (UNEP 2009) or if energy crops are rotated with food crops as done in some parts of Brazil (Moreira 2006).

4.1.1.1 Tropical ecosystems

Biofuels can be produced with the greatest efficiency in the tropics, and the lack of economic incentives for the conservation of tropical ecosystems means that they are vulnerable to replacement with biofuel crops (Doornbosch & Steenblik 2007). Indeed, the main tropical biofuel growing countries, Brazil, Indonesia and Malaysia, already have significant land use pressures and, as major deforestation 'hotspots', have been estimated to account for over half of forest clearance globally (Hansen *et al.* 2008). They also overlap with biodiversity hotspots (Koh 2007). It has been estimated that 540 Mha (Miles *et al.* 2008) to 745 Mha (Stickler *et al.* 2007) of tropical forests are suitable for oil palm or soybean production.

The expansion of oil palm plantations in Indonesia and Malaysia (which together account for 86% of global palm oil production) is the most cited example of forest loss for biofuel production[3]. It has been suggested that palm oil production is one of

[3] It should be noted that only a small proportion of total palm oil production is used as biofuel. However, projections suggest that demand for biofuels will become an increasingly significant driver of global oil seed production (OECD/FAO 2008)

the main drivers of increased deforestation rates across these countries (Turner & Foster 2009; Soyka *et al.* 2007), with an estimated 27% of oil palm concessions displacing peatland rainforest (Hoojier *et al.* 2006). More recent studies have estimated that 55-59% of oil palm expansion in Malaysia, and 56% in Indonesia occurred at the expense of forests (Koh & Wilcove 2007a), with others placing the figure as high as 80% in Malaysia (FAO 2008a). Although it is difficult to directly attribute forest loss to biofuel plantations, the 13% annual increase of the 3.6 Mha under oil palm in Indonesia has coincided with the annual loss of 1.8 Mha (2%) of the forest cover (FAO 2008a), and more increases are planned. Indeed, it has been estimated that 50% of the location permits issued for planned oil palm developments are on peatlands (Sheil *et al.* 2009), and that in Kalimantan, a number of threatened mammal species are found within areas of planned oil palm developments; which could consequentially have their ranges threatened (Venter *et al.* 2009). One simulation of oil palm plantation development, where permits are issued for forest conversion, suggests a loss of 20% of the primary and primary logged forest over 40 years (Sandker *et al.* 2007). Because Indonesia contains some three-quarters of Southeast Asia's remaining primary forests, the continuing loss of its primary forests would be disastrous for the region's biodiversity (Koh & Wilcove 2007b).

Expansion of biofuels onto natural ecosystems is not just a problem in Southeast Asia. In Latin America, for example, it has been reported that sugarcane is encroaching heavily into the Brazilian Cerrado, the world's most biodiverse savannah, and the Pantanal wetlands (Sawyer 2008). Soy is increasingly being used to make biodiesel, and recent increases in the conversion of the Cerrado and Amazon rainforests have closely tracked the expansion of soy plantations (Morton *et al.* 2008; Nepstad *et al.* 2008). A recent study has reported that 20% of the areas of high biological importance identified by the Brazilian Ministry of Environment are in the Cerrado, 70% of which overlap with potential areas for sugarcane expansion (Bustamante *et al.* 2009) but are not currently protected (Sawyer 2008). One study has estimated that two thirds of the Cerrado have been destroyed or degraded, threatening hundreds of species (Brown 2008).

Palm oil production has also been linked to large-scale deforestation in countries such as Colombia, Ecuador, Brazil, Central America, Uganda, and Cameroon, amongst others (AEA Technology 2008). In Africa, it has been suggested that sugarcane could become a major threat to biodiversity in tropical wetlands, and it has been reported that up to 550,000 ha of plantations are planned on river deltas in Kenya and Tanzania (Sielhorst *et al.* 2008).

4.1.1.2 Temperate ecosystems

In addition to land clearance in tropical areas, biofuel plantations are expanding the agricultural frontier in the US and Europe, particularly into subsidised set-aside land on which agricultural production is currently not permitted. Grassland set-aside for conservation has been converted for corn production in the USA, and similar patterns are seen for set-aside land in Europe (Meyerson 2008; Tilman *et al.* 2006). The European Environment Agency is concerned that the profitability of biofuels will outweigh the incentives for farmers to participate in agri-environmental schemes (House of Commons Environmental Audit Committee 2008). This is a significant issue for biodiversity, particularly for bird species (Scharlemann 2008), as it has been estimated that set-aside lands in the US have bird nesting rates that are ten times higher than those on cropland (Farrand & Ryan 2005). In Europe, encroachment into set-aside land by oilseed rape and sugar beet threatens semi-natural steppes and long-fallow dry cereal systems, which are among Europe's most biodiverse habitats. This has been linked to a decline in key habitats for endangered species (WWF 2006), and to declines in farmland birds (Brown 2008). A recent study of the potential biodiversity impacts of biofuel policy in Europe has suggested that although impacts will vary spatially and depend upon crop choice, they will be negative across all taxa, and particularly so for mammals (Eggers *et al.* 2009).

4.1.2 Indirect land use change in natural and semi-natural ecosystems

Indirect land use change (iLUC) occurs when the use of agricultural land for biofuels pushes other agricultural production into natural ecosystems. The inability to account for this displaced agricultural conversion means that the full impacts of biofuel production are largely unknown (Gallagher 2008). Sugarcane cultivation, for example, does not directly result in the loss of tropical forest but may replace pasture land or soy plantations, forcing expansion of livestock and soy production into the Amazon (Moreira 2006; Smeets *et al.* 2007; Sawyer 2008; Martinelli & Filoso 2008). Similarly, the increasing production of maize for ethanol in the US at the expense of domestic soy production has had the reported consequence of increasing soy expansion in the Amazon (Gallagher 2008).

The amount of iLUC is extremely difficult to quantify which makes it difficult to determine the impacts on biodiversity. However, if natural and semi-natural ecosystems are indirectly converted, negative impacts on biodiversity are likely to ensue (Hennenberg *et al.* 2009). Determining iLUC is particularily difficult where biofuel targets in one country are met through production in another country. For example, the US has specified that a large proportion of their renewable energy target will be met by 'advanced technology' biofuel, which can include sugar-based ethanol. As the current potential for second-generation biofuel is unclear, it is likely that the US will need to import sugarcane-based ethanol from Brazil to meet targets, potentially increasing feedstock expansion in the Brazilian Cerrado and displacing other crops into Amazon regions (Nepstad et al. 2008). However, new Brazilian law is being developed that prohibits sugarcane plantations on ecologically sensitive areas, which may result in other crops, such as soybean, expanding to these areas instead (Martinelli and Filoso 2008).

4.2 Biodiversity impacts of land conversion

Although the loss of natural ecosystems from biofuel production is often reported, the levels of biodiversity loss cannot be estimated without knowledge of the biodiversity present in the converted ecosystem. The few studies exploring the differences in species richness supported by biofuel feedstock plantations and natural ecosystems have mostly addressed oil palm, and have clearly shown that plantations support significantly lower levels of biodiversity than natural forest (Koh & Wilcove 2007b).

A recent review has identified only 13 publications documenting the impacts of oil palm plantations on animals, none for plants (Fitzherbert *et al.* 2008). It has been reported that biofuel plantations are generally intensively managed monocultures, have low species diversity and can also increase soil erosion. Recent studies have estimated that only 15% (Fitzherbert *et al.* 2008) to 22% (Danielsen *et al.* 2009) of species recorded in primary forest are found in oil palm plantations, and it has even been reported that oil palm plantations support lower levels of biodiversity than other tree crops, agricultural crops and abandoned pasture (Danielsen *et al.* 2009; Fitzherbert *et al.* 2008). For example, Turner & Foster (2009) reported a 72% reduction of arthropods in oil palm compared with primary forest, and Maddox *et al.* (2007) found that 32 of the 38 medium to large mammals occurring within forest sites in Sumatra were absent from oil palm.

Although oil palm plantations do support high levels of species richness for moths, ants and other insects, they do not tend to support forest species (Danielsen *et al.* 2009). A study in Sabah, Malaysia, found that species richness of the forest ground ant community was significantly lower in oil palm plantations than in the forest interior. Plantations supported only 5% of forest species, were dominated by non-forest species, and were found to be acting as dispersal barriers and leading to community isolation in rain forest remnants (Brühl & Eltz 2009).

It has also been reported that the species lost when forest is converted to oil palm tend to be specialist species of highest conservation concern (Fitzherbert *et al.* 2008). For example, Aratrakorn *et al.* (2006) found that oil palm plantations in Thailand not only supported fewer bird species than forest, but that these species were significantly more widespread and of lower conservation concern than those in forest. Similar results have been found in riparian forest fragments of sugarcane-dominated watershed, where an increase of generalist species such as the capybara that are typical of degraded areas have been recorded, along with an absence of mammals typical for intact forest (Martinelli & Filoso 2008). Further, it has been reported that many of the species recorded in abundance in oil palm monocultures are invasive, and replace species important for ecosystem functioning such as pollinators (Fitzherbert *et al.* 2008).

As well as forest loss, biofuel plantations can result in forest fragmentation. It has been reported that forest fragments in oil palm plantation areas support less than half the species of continuous forest, and suffer tree sapling mortality from wild pig populations that forage in the plantations (Fitzherbert *et al.* 2008). Oil palm plantations have fragmented and destroyed the habitat of the Sumatran rhino, Asian elephants, and the tiger, increasing human-wildlife conflict (Uryu *et al.* 2008). Moreover, the burning of land to create oil palm plantations in South East Asia is thought to have contributed to the forest fires of 1997 that killed an estimated one third of Borneo's orang-utan population (Sandker *et al.* 2007). The development of infrastructure in previously inaccessible areas is also a risk to biodiversity (Cramer 2007).

Most of the studies found relate to oil palm but there are many other feedstocks. The impacts of these feedstocks on biodiversity due to land conversion will ultimately depend on the previous land use. Conversion of degraded or marginal land may result in increased biodiversity (see below).

4.3 Conversion of degraded or marginal lands

When appropriate crops are planted in suitable areas, they can actually benefit biodiversity (Farrell et al. 2006). This is particularly true where biofuels are grown on marginal and degraded lands (Eickhout *et al.* 2008). They can increase soil productivity, reduce soil erosion, reduce pressure on natural ecosystems, and create habitats (Tilman *et al.* 2006). Jatropha is one biofuel crop that can be grown on degraded land and is receiving increasing attention, particularly in Africa and India (Grain 2007). It has been suggested that this crop can be grown by traditional pastoralists under traditional systems that maintain biodiversity (Lovett 2007), and it is not a crop used for food production. However, its land and water requirements and suitability for large scale production are yet to be fully determined, and reports suggest that it will grow better on more productive land (AEA Technology 2008; Eggers *et al.* 2009), meaning that production on degraded land would need to be incentivized. On the other hand, some countries currently have biofuel crops on non-sensitive lands (Machado-Filho 2008) or still have large expanses of such land available for energy crop production (UNCTAD 2006).

4.4 Post land conversion impacts

There has been very little assessment of the post land conversion impacts of the plantations themselves. As such, the true environmental impacts of biofuels are often overlooked (Groom *et al.* 2008; Turner *et al.* 2008). Life cycle analyses (LCAs) provide a mechanism for these impacts to be assessed. Along with greenhouse gas emission, LCA provides a mechanism for investigating biofuels' potential for acidification, eutrophication, toxicity, photochemical ozone, and ozone and resource depletion. However, few LCAs provide a comprehensive coverage of potential environmental impacts and impacts on biodiversity are not covered due to a lack of indicators (Menichetti & Otto 2009).

Although overlooked, the pollution from fertilisers and pesticides is likely to be another major negative biodiversity impact associated with biofuels; particularly for aquatic ecosystems (Sala *et al.* 2009; Johnston *et al.* 2009). One study on the environmental impact of sugarcane plantations has shown that soil erosion is high in plantations in comparison to forest and pasture, and that the resulting sediments are deposited into wetlands, rivers, and streams (Martinelli & Filoso 2008). The same study noted that watersheds in major sugarcane areas had only 13-18% of the original riparian vegetation, which has led to decreasing small mammal species richness (Martinelli & Filoso 2008). However, the cultivation of appropriate native species could result in less pollution, water stress and provide wildlife habitats (UN-Energy 2007).

The environmental impacts of corn-based ethanol (a fertiliser and pesticide intensive crop) are thought to be the highest of any agricultural crops in the US (Groom *et al.* 2008). Again, many of these impacts are felt downstream, making it difficult to quantify. For example, ethanol production in the Mississippi River system has been linked to hypoxia in the Gulf of Mexico (Sala *et al.* 2009). In addition, biofuel crops can cause water stress. Many of the crops require heavy irrigation, which can involve the drainage of wetlands and use of water from rivers and lakes, with consequent implications for biodiversity (AEA Technology 2008). Despite this, in-depth studies into the water footprints of different biofuel crops are only just becoming available (Gerbens-Leenes *et al.* 2009). There is also significant concern that some biofuel crops will be invasive in some areas. For example, a study in Hawaii has suggested that 70% of regionally suitable biofuel crops have a high risk of becoming invasive, compared to only one quarter of the non-biofuel plant species assessed (Buddenhagen *et al.* 2009).

Biofuels that do not require high-energy inputs and can be grown in polycultures of native species will be more biodiversity friendly than monocultures (Tilman *et al.* 2006). This does not apply to most of the first generation biofuels, but it has been suggested that the development of second-generation biofuels such as grassland perennials could potentially lead to more 'biodiversity friendly' biofuel production monocultures (Tilman *et al.* 2006). Choice of crop and management regime is therefore an important element in determining whether impacts on biodiversity are positive or negative.

The ‘off farm’ impacts of biofuel production will be similar to those observed from conventional agriculture, a full analysis of which is beyond the scope of this report. These impacts have been reviewed and related to biofuels in a report by AEA Technology for the UK government (AEA Technology 2008).

4.5 Influence of management regimes on biodiversity

Management can greatly influence the extent of the impacts of biofuels on biodiversity. There is some evidence, for example, that well-managed oil palm plantations can provide a water regulation service, although more work is required in this area (Sheil *et al.* 2009).

Most of the examples in the literature to date are focused specifically on (the negative impacts of) oil palm plantations, but there are a number of different biofuel crops that provide different amounts of energy and have varying environmental impacts (Royal Society 2008; Scharlemann & Laurance 2008b). In addition, the same biofuel crop can have vastly different impacts depending on where it is planted and the previous land use as well as the management of the area. There is wide variation in agricultural conditions, soil types, water availability and climatic conditions between producer countries, all of which are important for determining environmental (AEA Technology 2008). Given that biofuel production is likely to increase, an understanding of how and if biofuel plantations can be managed to limit negative impacts on biodiversity, and even be beneficial for biodiversity, would seem vital.

A number of good practice examples have been identified in the biofuels literature. Avoiding use of fire can avoid negative impacts on seeds and sedentary animals, and integrated pest management and limited fertilizer applications will reduce negative impacts such as water pollution (Fitzherbert *et al.* 2008). Sustainable management practices such as no-till cultivation and the use of cover crops can all improve environmental performance (Hill *et al.* 2006). The maintenance of forest fragments, buffers and corridors to provide stepping stones for species movements, or the management of plantations as agroforestry systems (where trees are interspersed with crops) (Bhagwat & Willis 2009; Koh 2008), have all been suggested as options to reduce biodiversity impacts of biofuel feedstock plantations. These practices are already being employed in some cases. Sao Paulo state for example requires 20% of plantation area to be set as natural reserves and will ban burning of sugarcane fields prior to harvest (Worldwatch Institute 2007).

Some authors stress that it is difficult to enhance the biodiversity value of plantations, and that there is a trade-off between reducing management intensity and minimizing land use requirements (Phalan *et al.* 2009). There are some who advocate focusing conservation efforts on reducing deforestation rather than preventing monoculture plantations (Phalan *et al.* 2009), as where natural ecosystems are replaced with biofuel the impacts will be negative whatever management practices are employed. Others suggest that the intensification of agricultural practices can be just as damaging when the impacts of eutrophication and soil nutrient loss are taken into account (Johnston *et al.* 2009). This is part of the ongoing debate over the biodiversity benefits of intensive vs. extensive agriculture (Green *et al.* 2005); a full assessment of which is beyond the scope of this report.

Due to the initial impacts of land conversion, policies to promote careful land use planning, such as the setting of sustainability criteria, are likely to go furthest to reduce the negative biodiversity impacts (House of Commons Environmental Audit Committee 2008), and will be discussed in section 6. It has been suggested that careful landscape planning within the agricultural mosaic, taking into account heterogeneity, connectivity, and land suitability could improve the environmental performance of biofuels (Milder *et al.* 2008).

5 Biodiversity impacts of 'next generation' biofuels

5.1 Second generation

Much of the hope for future global biofuel production is pinned on the large-scale development of 'second generation' biofuels (Robertson *et al.* 2008; Farrell *et al.* 2006; Doornbosch & Steenblik 2007). This involves the use of advanced technology to break down lignocellulosic plant matter into sugars, which can then be distilled to ethanol fuel, or the gasification of biomass followed by conversion of gas to liquid (the Fischer – Troper process). Lignocellulosic bioenergy crops are typically woody and herbaceous perennial species (Hill *et al.* 2007). Switchgrass, *Miscanthus*, wood plantations (e.g. poplar), as well as forestry and agricultural waste products may all be converted to biofuel using these advanced technologies. These processes typically yield more energy than traditional biofuel (Heaton *et al.* 2008), but large-scale production is still under development.

5.1.1 Reducing land conversion

There is some disagreement in the literature over the potential for second generation biofuels to reduce the negative biodiversity impacts of biofuel production. In theory, second generation biofuels could remove some of the issues with land conversion, as they can be grown on marginal land (Schmer *et al.* 2008), and can be produced from agricultural and forestry waste products (Koh & Ghazoul 2008). In such cases, feedstock cultivation would either have no additional impact or provide some benefits with regards to biodiversity.

There is some debate regarding the amount of land needed for the second generation of biofuels. For example, Somerville (2007) claims that *Miscanthus* would require only 3.2% of the terrestrial surface area to meet global energy needs, and its potential as an energy source has been recognized by a number of authors (Heaton *et al.* 2008).

Further, it has been estimated that one quarter of current energy demand could be met using waste residues from agriculture and forestry (Junginger *et al.* 2006), which have scored highly on environmental performance in comparison to other biofuel sources (Zah *et al.* 2007). However, other studies have suggested that second generation biofuels may require a larger land area (on a global scale) than first generation biofuels (Gallagher 2008; Gurgel *et al.* 2008; Rubin 2008; FAO 2008b). This is largely due to the fact that second generation biofuels do not produce beneficial co-products such as animal fodder, which would need to be grown separately (Eickhout *et al.* 2008; Farrell *et al.* 2006; Gallagher 2008). The potential of 'second generation' biofuel for climate change mitigation could also be considered doubtful if it involves the large areas of land use change projected (Gallagher 2008).

Again, the biodiversity impact will largely depend upon the previous land use. Although planting second generation biofuels on degraded land could reduce natural land conversion whilst providing soil stability, nutrients, and increasing water retention (Biemans *et al.* 2008; Schmer *et al.* 2008), it has been estimated that yields of grassland perennials are 50% higher if grown on fertile land (Tilman *et al.* 2006). Plantations on degraded land would therefore have to be properly incentivised, and it has been suggested that regulations would be required to stop crops such as switchgrass being grown like traditional biofuels (Schmer *et al.* 2008). Land compatible with switchgrass is the same as that currently set-aside under the USDA Conservation Reserve Programme, and it is likely that biodiverse 'prairies', pasture land and cropland would be required to meet biofuel targets (Perlack *et al.* 2005; Schmer *et al.* 2008). Similarly, incentives for short rotation coppice (SRC) could result in the conversion of forest to poplar and willow plantations (Groom *et al.* 2008).

5.1.2 Post land conversion impacts

Aside from their ability to grow on degraded land, it has been suggested that second generation biofuel plantations can be more biodiversity friendly (Tilman *et al.* 2006). Perennial crops can reduce the need for fertilisers and tillage, and can be grown as species mixtures (Robertson *et al.* 2008). Recent research has suggested that native prairie species grown on 'degraded land' can produce 238% more energy than monocultures with little fertiliser input, whilst providing biodiversity benefits through habitat creation, prevention of soil erosion and nitrogen fixation (Tilman *et al.* 2006). It has been found, for example, that SRC poplar and willow in the UK supported higher species diversity than agricultural controls, although it was noted that such plantations were not a substitute for natural land (Rowe *et al.* 2009). Moreover, a recent study has found them to perform badly in life cycle analyses (Jungbluth *et al.* 2008). Woody biofuels can also provide an incentive for better forest management practices and provide ecosystem services if planted on degraded land (Biemans *et al.* 2008; Eickhout *et al.* 2008; Royal Society 2008). It has also been suggested that planting biomass crops on annual crop lands could improve land ecology (Khosla 2008).

However, recent reports have questioned the capacity of second generation biofuels to reduce the negative biodiversity impacts for a number of reasons. Although second generation biofuels have the potential to be grown in species mixes, the potential for such production on a large scale is not known, as it is unclear how they would be processed (Perlack *et al.* 2005). Monoculture plantations are still likely to be more financially viable, and would have similar impacts to those of first generation biofuels if established in natural areas (Biemans *et al.* 2008). Some cellulosic crops require significant chemical input, and have high environmental impacts when the full life cycle analysis is considered (Jungbluth *et al.* 2008). Further, where all 'waste' biomass is removed for fuel this can impact on soil fertility and cause erosion (Eickhout *et al.* 2008; Gallagher 2008; Schmer *et al.* 2008). One study in the US has estimated that 1.3 billion tonnes of dry biomass would be required to provide energy requirements, resulting in the removal of most residues from soil and a need for increased fertiliser use (Perlack *et al.* 2005). Corn stover, for example, provides essential plant nutrients and buffers against soil erosion, and negative impacts have been shown on soils where more than 25% of the stover is removed (Blanco-Canqui & Lal 2009**.** Similarly, an estimated 25% of all woodland species depend on 'forestry waste' (Perlack *et al.* 2005), and deadwood is particularly important for biodiversity and productivity in boreal ecosystems (Biemans *et al.* 2008). However, waste products from saw mills and the use of grass cuttings is not likely to negatively impact biodiversity (Biemans *et al.* 2008).

Another major concern is that monoculture non-food crops such as switchgrass have invasive traits and could have significant negative biodiversity impacts if they spread into natural ecosystems (Meyerson 2008; Royal Society 2008; Sala *et al.* 2009). As with first generation biofuels, the potential to reduce negative impacts will depend on the crop produced, where it is produced, and the method of production; and careful analysis is required (Robertson *et al.* 2008). The potential impacts of large scale production are largely unknown (Rowe *et al.* 2009).

5.1.3 Potential for GM organisms

One option to increase production of biofuel is through the use of genetically modified (GM) biofuel crops. Genetic modification techniques can include the modification of plants to make the lignin easier to break down, and the speeding up of growth and yield. This can apply to agricultural, grass, and tree crops (Global Forest Coalition & Global Justice Ecology Project 2008). The use of genetically modified biofuel crops carries risks for biodiversity, ranging from the potential pollen transfer to wild crop relatives to increased agrochemical application. The risk of gene transfer to natural ecosystems is a particular risk for tree species, which are long lived and spread seeds widely (Global Forest Coalition & Global Justice Ecology Project 2008).

Conversely, if GM biofuel crops can be produced more efficiently than conventional crops this could reduce the land required for biofuel production, thus reducing the direct and indirect conversion of natural ecosystems. The CBD recommends the use of the precautionary principle when making decisions on GM crops and trees (UNEP/CBD/COP/DEC/IX/5).

5.1.4 Algal (third generation) biofuels

The literature on algal biofuels is still very limited, and climate change mitigation potential of algal biodiesel remains to be seen, but there is some optimism (Jenner 2008; Wang *et al.* 2008). Microalgal biodiesel has high energy potential, as most of the algal dry weight can be used in production (Patil *et al.* 2008; Chisti 2008; Herro 2008), and has been cited as the only renewable biofuel source that has the potential to completely displace petroleum-derived transport fuels (Chisti 2008). It has been estimated that microalgae could account for half of the transport fuel needs of the US with just 1.1% of the country's cropland (Chisti 2008), and it can reportedly produce more fuel per area of land than maize, rapeseed or jatropha even when grown on land that is not suitable for agriculture, in seawater and brackish water (Gross 2008). If this were the case, then there would be significant potential to reduce biodiversity impacts, but there does not appear to have been any investigation of algal biofuels in this context. Currently the barriers to production appear to be economic, and large-scale production does not appear likely over the short term.

6 Sustainability criteria and their potential to reduce negative impacts on biodiversity

Several initiatives by governments or international bodies, such as the Roundtable on Sustainable Biofuels, have developed or are developing 'sustainability' standards and criteria for biofuel production. These standards, which can either be regulatory or voluntary, typically include criteria designed to ensure greenhouse gas savings, limit production of biofuels on carbon rich or biodiverse areas, and promote good environmental management, food security and social justice.

Regulatory standards are set by governing bodies, and adherence to them is mandatory. The European Commission (EC), for example, has developed regulatory standards through the renewable energy sources directive (EU-RES-D) that apply to all biofuel feedstocks used to meet the renewable energy targets, whether grown in or imported to the EU (Hennenberg *et al.* 2009). Switzerland has set stringent standards where all biofuels derived from oil palm, corn and soybeans are banned outright, and all other biofuels can only be used after they are evaluated against a range of criteria addressing GHG emissions, social, and environmental performance (Searchinger 2009).

Voluntary standards are developed by industry or other interested groups such as non-governmental organisations (NGOs), and typically set out criteria or principles that producers can adhere to in order to get accreditation to that standard. They lack the legal clout of regulatory standards, but can be applied across a wider geographic area. Initiatives such as the Roundtable on Sustainable Biofuels (RSB), the Better Sugar Cane Initiative, the Roundtable on Responsible Soy (RTRS) and the Roundtable on Sustainable Palm Oil (RSPO) have developed or are in the process of developing voluntary standards that consider, amongst other things, the biodiversity impacts of biofuel production (Palmujoki 2009). A full list and comparison of these initiatives can be found in Hennenberg *et al.* (2009).

To limit adverse biodiversity impacts, standards typically aim to influence where biofuels are produced, rather than how they are produced. They try to do this through the designation of certain ecosystems and areas of high importance for biodiversity as ‘no go’ areas for biofuel production, and the recommendation of areas suitable for biofuel production such as ‘degraded land’ (Fritsche *et al.* 2008). However, a number of practical issues are still to be resolved, including the contentious issues of how to define areas of ‘high biodiversity importance’ and ‘degraded’ land. Other issues include the incorporation of iLUC, and the implementation and verification of these criteria on the ground.

6.1 Areas of high biodiversity importance

A recent review of sustainability criteria found that all of the initiatives assessed included criteria related to both areas of high biodiversity and endangered or vulnerable species, as well as language to avoid the conversion of natural habitats and particular ecosystems such as forest (Hennenberg *et al.* 2009). However, ‘high biodiversity importance’ is difficult to define (Nelson & Robertson 2008). An area may be of importance for biodiversity if it supports high species diversity, or if it supports species, habitats and ecosystems of conservation concern. Further, as not all biodiversity is fixed to a location there may be a temporal element in the importance of an area (e.g. sites of aggregation of migratory species). Conversion of protected areas is usually explicitly prohibited by sustainability standards, but many of these areas of high biodiversity importance fall outside of the protected area network (Hennenberg *et al.* 2009). However, the wide range of other ways in which particular locations can be significant for biodiversity means that identifying areas of high biodiversity importance can be problematic and very dependent on the scale at which an assessment is made.

At the global scale, conservation scientists have used several different approaches to identify areas of importance for biodiversity conservation, such as Conservation International’s biodiversity hotspots, or WWF’s global 200 ecoregions, but these are generally not considered appropriate for decision making at the scale of biofuel production. At the national scale, Key Biodiversity Areas (KBAs) identify important sites for biodiversity (particularly for endangered and endemic species) and include sites that have no legal protection (Eken *et al.* 2004). The High Conservation Value (HCV) concept is used in a number of sustainability criteria (Dehue *et al.* 2007), and recognises that areas have different types of conservation value that need to be defined in national and local contexts. HCV areas are identified through consultation, and are identified and mapped according to six loosely defined HCV categories of important land use types (Stewart *et al.* 2008). The RSB, for example, states that biofuel production should 'avoid negative impacts on biodiversity, ecosystems, and areas of High Conservation Value', and provides guidance that HCV areas can only be used in biofuel production if conservation values are left intact (RSB 2008). Other standards completely prohibit any exploitation or conversion of these areas for feedstock production (Hennenberg *et al.* 2009). There is little discussion in the literature of the relationship between the various standards and their varying levels of protection for 'high biodiversity' lands but it is clear that there is little consensus on how they should be defined and identified, and the identification of HCV lands is open to interpretation.

In addition to the high biodiversity areas, sustainability standards often include provisions to reduce the conversion of natural or semi-natural ecosystems; the loss of which could have significant negative impacts on biodiversity (Hennenberg *et al.* 2009). The EC Directive, for example, excludes biofuel production in 'pristine forests', 'high biodiversity grasslands', 'wetlands' and 'high carbon areas', and the RSB restricts conversion of native ecosystems and ecological corridors (RSB 2008). Whilst these provisions would deliver significant safeguards for biodiversity if properly applied, the definitions of these terms are still to be developed and are open to interpretation (Searchinger 2009). In Indonesia, for example, the exemption of only pristine forests would allow the conversion of the millions of acres of logged forest, which have high biodiversity value (Sheil *et al.* 2009). Similarly, it is unclear exactly what constitutes 'high biodiversity grassland', and this will differ from country to country according to relative biodiversity levels and existing management practices.

6.2 Degraded land

Most sustainability criteria promote the production of biofuels on 'degraded land' to avoid impacts on food production and biodiversity. Several recent studies have suggested that expanding biofuels into degraded land could achieve carbon savings whilst significantly reducing biodiversity impacts (Metzger & Hutterman 2008; Bindraban *et al.* 2009; Fargione *et al.* 2008; Gibbs *et al.* 2008), and some authors have recommended that the production of biofuel feedstocks should be limited to degraded land and waste products (Gallagher 2008).

The problem with this concept is that there is no accepted definition of 'degraded land' (RSC 2008). This has led to vastly different estimates of the global availability of such land. Recent research has suggested that the global potential for bioenergy production on abandoned agricultural land is 5% (Field *et al.* 2008) to 8% (Campbell *et al.* 2008) of current energy demand globally, and will therefore be insufficient to meet even the targets of the EU and US (Kanter 2008). The estimate by Campbell *et al.* (2008), which is based on historical land use data, satellite derived land cover data and global ecosystem modelling, provides a range of 385-472 Mha of available abandoned agricultural land, which does not compare favourably to most of the estimates of land requirements for biofuel production reported in Section 3. Land availability estimates very rarely take into account the land requirements for other purposes such as afforestation and renewable energy.

It has been suggested that taking ‘degraded land’ to mean ‘low carbon land’ could have significant biodiversity risks. For example, the Brazilian Cerrado is low in carbon compared with other lands (Fargione *et al.* 2008) and strictly following carbon-related sustainability criteria would justify its conversion although it is high in biodiversity. Even abandoned agricultural lands may have significant biodiversity importance (Ravindranath *et al.* 2009). Some of the ‘degraded’ lands of Europe or North America have been set aside for conservation and provide important habitat for birds such as Eurasian Skylarks (Scharlemann 2008). Under some definitions, logged forests could be considered degraded and available for biofuel production (Koh & Wilcove 2009) despite estimates that forests in Malaysia and Indonesia can recover 84% of their bird species 30 years after selective logging, whereas only 27% of bird species and 21% of butterfly species are estimated to remain after conversion of logged forests to oil palm (Koh & Wilcove, 2007a).

There are concerns that such definitions of degraded land could create incentives for conversion of degraded but high biodiversity forest, particularly as degraded forests have already been earmarked for conversion in some areas of Asia (FAO 2008a). Similar concerns have been raised over the status of semi-natural grasslands, which can contain high levels of biodiversity but are often overlooked (Ceotto 2008; Lysen *et al.* 2008; EEA 2004). It would, therefore, appear that the status of these ‘degraded lands’ needs to be carefully evaluated (Fritsche *et al.* 2008), with thorough evaluations that identify any potential negative impacts and put measures in place to mitigate them (Hennenberg *et al.* 2009).

Regardless of land availability concerns, biofuel production on degraded land is economically inferior to production on higher quality agricultural land. In Malaysia, most oil palm plantations are on newly cleared forest land, despite the existence of abandoned land (Nellemann *et al.* 2007). Even though second generation biofuels can be grown on marginal lands, they will have high water and fertiliser requirements, and yields will always be higher on productive land (Field *et al.* 2007). In addition, timber profits gained from land clearing can be used to overcome the initial costs of plantation development (EAC, 2008); a source of income that is not available from degraded land. Therefore, production on degraded land is unlikely to be implemented unless the standards imposed create economic incentives (AEA Technology 2008).

6.3 Indirect land use change

The limited literature on this topic suggests that none of the current standards or environmental impact assessments (e.g. LCA; Menichette and Otto 2009) adequately addresses the issue of indirect land use change (iLUC) (Hennenberg *et al.* 2009). Furthermore it has been suggested that sustainability standards will not be able to fully address biodiversity impacts unless they can ensure that the displacement of land use does not affect highly biodiverse areas (Doornbosch & Steenblik 2007). Although it is thought that this issue will be at least partially resolved through the production of biofuels on degraded land or by using waste products (Gallagher 2008; Fritsche *et al.* 2008), others suggest that it will be difficult to overcome this issue without a full land use planning or accounting approach that incorporates agricultural and conservation (Hennenberg *et al.* 2009), and greater transparency in land use decisions (Koh & Wilcove 2009). Indeed, although there are models that aim at assessing indirect land-use change, assessment is usually limited to displacement of agricultural crops. However, these crops may displace 'degraded' pasture which may in turn displace biodiverse land (Ceotto 2008; Moreira 2007). Furthermore, even where effective land use policies are effective at addressing iLUC in one area, they will not be avoided unless they are adopted globally and for all agricultural commodities (Hennenberg *et al.* 2009). Assessing iLUC is clearly difficult and is to a large extent an uncertain process, which requires further research (Cornelissen and Dehue 2009).

6.4 Practical implementation of standards

Although much of the literature focuses on the provisions included within biofuel standards, the success of any given standard will depend upon both the interpretation and application of that standard when implemented by the relevant stakeholders on the ground (Hennenberg). Some stakeholders have expressed concern that the standards are complicated and costly to implement, and that few independent assessments of their effectiveness have been made (Sheil *et al.* 2009). In addition, voluntary standards will only be effective if a large number of producers conform to them, and regulatory standards will only be effective if properly applied and enforced (House of Commons Environmental Audit Committee 2008). There are also concerns that the large number of different certification schemes will limit credibility and meaningful participation of stakeholders (Kaphengst *et al.* 2009; Zarrilli 2008). Finally, a particular omission identified in the literature is a full understanding of the environmental impacts of each biofuel feedstock (Doornbosch & Steenblik 2007), which would enable the identification of appropriate cultivation systems and best management practices (Hennenberg *et al.* 2009).

It has been suggested that lessons should be learned from forestry certification (Doornbosch & Steenblik 2007). Sustainability standards may help to reduce the negative biodiversity impacts of biofuels, but it appears that more work is needed to ensure that these standards will achieve their goals. Issues identified in the literature include a lack of clarity on definitions, the lack of measures to prevent iLUC, and the confusing array of standards. It has also been emphasized that sustainability criteria are only part of the required solution (Kaphengst *et al.* 2009), and that decision makers will need to understand the implications of policy measures and management practices at different spatial scales in the context of wider land use planning (Robertson *et al.* 2008).

7 Conclusion

The production of liquid biofuels is rapidly increasing. Demand-based projections suggest that this trend is likely to continue, largely driven by governmental targets and subsidies.

The impacts on biodiversity will depend upon the biofuel feedstocks, previous land use, and agricultural practices employed, and can be positive where well-managed plantations are established in suitable areas. However, there is evidence that the cultivation of many of the biofuel feedstocks are already having negative impacts on biodiversity as a result of habitat conversion and the 'off-farm' impacts of pollution and soil erosion. Most concern oil palm plantations, however. Further negative impacts are likely to be observed in the future as the land requirements for biofuel feedstock production increase. Indeed, the limited literature available suggests that biodiversity will continue to be negatively impacted under most current scenarios of biofuel production, largely as a result of habitat loss and fragmentation. The development of 'next generation' biofuels offers some potential for reducing biodiversity impacts, as perennial species grown on marginal lands and waste products from agriculture and forestry can be utilised. However, the potential impacts of large-scale production are largely unknown, and there is some scepticism over their ability to reduce land use requirements. There are also concerns over the use of invasive species, and the removal of 'waste' products from soil.

Given that biofuel production is increasing, a comprehensive assessment of the environmental impacts of biofuel production, and the identification of measures to reduce these impacts, is required at local to regional scales. Sustainability standards for biofuel production may help to reduce adverse impacts on biodiversity, and a number of these are currently under development, or in the early stages of implementation. However, they will need to overcome a number of issues surrounding definitions of key terms, and address the issue of indirect land use change if they are to be successfully implemented. In addition, it is likely that sustainability standards will only be part of the solution, and will need to be combined with improved land use planning.

Biofuels have the potential to contribute to climate change mitigation. However, this may need to be balanced against the negative impacts on biodiversity. The impacts on biodiversity are not always obvious (e.g. from indirect land use change) and more research is needed, especially at the local level since much of the current literature reviewed focuses on global overviews.

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