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The impact of biomass crop cultivation on temperate biodiversity

JENS DAUBER, MICHAEL B. JONES and JANE C. STOUT
Department of Botany, School of Natural Sciences, Trinity College Dublin, Dublin 2, Ireland

Abstract

The urgency for mitigation actions in response to climate change has stimulated policy makers to encourage the rapid expansion of bioenergy, resulting in major land-use changes over short timescales. Despite the potential impacts on biodiversity and the environment, scientific concerns about large-scale bioenergy production have only recently been given adequate attention. Environmental standards or legislative provisions in the majority of countries are still lagging behind the rapid development of energy crops. Ranging from the field to the regional scale, this review (i) summarizes the current knowledge about the impact of biomass crops on biodiversity in temperate regions, (ii) identifies knowledge gaps and (iii) drafts guidelines for a sustainable biomass crop production with respect to biodiversity conservation. The majority of studies report positive effects on biodiversity at the field scale but impacts strongly depend on the management, age, size and heterogeneity of the biomass plantations. At the regional scale, significant uncertainties exist and there is a major concern that extensive commercial production could have negative effects on biodiversity, in particular in areas of high nature-conservation value. However, integration of biomass crops into agricultural landscapes could stimulate rural economy, thus counteracting negative impacts of farm abandonment or supporting restoration of degraded land, resulting in improved biodiversity values. Given the extent of landconversion necessary to reach the bioenergy targets, the spatial layout and distribution of biomass plantations will determine impacts. To ensure sustainable biomass crop production, biodiversity would therefore have to become an essential part of risk assessment measures in all those countries which have not yet committed to making it an obligatory part of strategic landscape planning. Integrated environmental and economic research is necessary to formulate standards that help support long-term economic and ecological sustainability of biomass production and avoid costly mistakes in our attempts to mitigate climate change.

Keywords: bioenergy, biomass crops, ecosystem services, land-use change, policy guidelines, short-rotation coppice, spatial scale dependency, sustainability

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Introduction

Energy crops are promoted as a promising renewable energy source that could reduce human dependence on fossil fuels and form an important component in a portfolio of climate mitigation measures, by both lowering greenhouse gas (GHG) emissions and sequestering carbon in soils (Farrell *et al*., 2006; Ragauskas *et al*., 2006; Sims *et al*., 2006). High expectations of energy crops have stimulated policy makers in Europe and North America to encourage their rapid expansion by subsidizing their production to meet future energy and environmental targets (Field *et al*., 2008; Groom *et al*., 2008; UNEP, 2009). However, the urgency for mitigating actions in response to climate change may have resulted in inadequate consideration of scientific concerns about

Correspondence: J. Dauber, tel. +353 1 896 3761, fax +353 1 896 1147, e-mail: jens.dauber@vti.bund.de

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the environmental and socio-economic sustainability of large-scale energy crop production, as well as its effectiveness for reaching energy security and GHG mitigation targets (e.g. Hill *et al*., 2006; Robertson *et al*., 2008; Russi, 2008; Searchinger *et al*., 2008; Florin & Bunting, 2009; Petrou & Pappis, 2009; Tilman *et al*., 2009). In order to reach the benchmarks of energy production from bioenergy, vast areas of land will have to be converted to energy crop production, resulting in major land-use changes over relatively short timescales (RCEP, 2004; EEA, 2006; Tuck *et al*., 2006; Marland & Obersteiner, 2008; Fischer *et al*., 2010). Despite the spatial extent of this development and the potential severity of its impact on the environment, energy crop planting in many countries is currently done without appraisal and with few environmental standards or legislative provisions delineating principles of energy crop production (UN-Energy, 2007; Groom *et al*., 2008; but see Haughton *et al*., 2009).

Land-use change, in general, is considered one of the major drivers of biodiversity loss (Sala *et al*., 2005). Therefore, although indirect positive effects of energy crop production on biodiversity through potential halting of climate change are acknowledged, there is concern about the direct effects that expansion of energy crop production could have on biodiversity (Huston & Marland, 2003; Robertson *et al*., 2008; Eggers *et al*., 2009). Already, the cultivation of energy crops has a significant impact on land use. Out of the total area of arable land in 25 European Union countries in 2005 (97 Mha), about 1.8 Mha were used for producing raw materials for bioenergy (Commission of the European Communities, 2005). At national levels, for instance in England, the area of land planted with biomass crops such as *Miscanthus* (predominantly *Miscanthus* × *giganteus*) and short rotation coppice (SRC) willow (*Salix* spp.) has increased more than seven times since 2003 to an estimated area of 15 000 ha (Haughton *et al*., 2009). The UK government's Biomass Strategy (Defra, 2007); however, suggests that the area occupied by bioenergy crops, grown for heat and power generation, could reach 1.1 million ha by 2020. Similarly, in Germany, the land area used for production of energy crops, in particular of maize for biogas, quadrupled during the 10 years to 2008, to cover ca. 17% of agricultural land (Wiehe *et al*., 2009).

There is, therefore, an urgent need for well-structured, conceptual frameworks that connect scientific knowledge with policy making and action, to establish environmental and biofuel certification standards for energy crop production (Groom *et al*., 2008; Meinke *et al*., 2009). Drafting such standards is a challenging task because the primary objectives of energy crop cultivation are climate change mitigation and energy security and not the support of wildlife-friendly farming systems. Nevertheless, preliminary studies show that economically viable and environmentally sustainable, integrated food and energy agro-ecosystems might be feasible (Porter *et al*., 2009). The wide scope for land-use planning which includes energy crops could present an opportunity for novel agricultural landscapes of higher economic viability and environmental sustainability, provided that holistic, knowledge-based planning concepts are developed and applied (Scherr & McNeely, 2008; Koh *et al*., 2009). Application of scientific expertise on how energy crop cultivation affects biodiversity, ecosystem services, economic viability and ecological sustainability of natural habitats and agricultural lands can contribute to evaluation and planning standards that help in achieving win–win solutions for biodiversity conservation, GHG control and energy security (Huston & Marland, 2003; Groom *et al*., 2008).

In this context it is important to differentiate between first and second generation energy crops as management intensity, implications for land-use change and, as a result, impacts on biodiversity differ among the crops involved (UN-Energy, 2007; UNEP, 2009). In contrast to first generation biofuels, which are currently grown as arable food crops rich in sugar, starch or vegetable oil (such as maize, soybean or rape seed), but can be used for ethanol and biodiesel production, second generation feedstock are perennial ligno-cellulosic crops such as fast growing trees (SRC) or grasses used for combustion or ethanol production (see Karp & Shield, 2008 for an overview). Compared with arable crops, ligno-cellulosic biomass crops have the potential for positive effects on soil carbon sequestration and soil properties in general, GHG emissions, biodiversity and energy balance (Styles & Jones, 2008; Rowe *et al*., 2009). The major focus of this review is on second generation biomass crops, in particular on SRC crops and perennial grasses, because they are currently considered to be the most efficient and sustainable feedstock for bioenergy production in temperate regions (Adler *et al*., 2007; Karp & Shield, 2008; Russi, 2008; Williams *et al*., 2009; UNEP, 2009).

The aims of this review are (i) to summarize the current knowledge about the impact of biomass crop production on biodiversity in temperate regions with an emphasis on the major producers, Europe and North America, (ii) to identify knowledge gaps and (iii) to draft guidelines for a sustainable energy crop production with respect to biodiversity conservation, covering a range of spatial and temporal scales.

The effects of biomass crop production on biodiversity and associated ecosystem services depend greatly on the respective crop and its management, how production is integrated into existing landscapes and farming systems, how much land is converted and whether intensively managed agricultural land, marginal agricultural land or natural areas are affected (e.g. Ranney & Mann, 1994). As a consequence, impacts of biomass crops on biodiversity operate on a wide range of spatial scales. In this review, we have separated impacts of biomass crops at the field/crop, landscape and regional scale because this distinction facilitates isolating particular pressures and impacts and how they might be assessed (Firbank, 2008). At the field scale, we look at the intrinsic biodiversity value of the biomass crops in comparison with the crops replaced and to alternative field usages. We review the intensity of management required for biomass crop cultivation, the structural habitat quality and effects of secondary land use of energy crops such as waste water disposal. At the landscape scale, we examine the spatial structure and turnover of habitats within a landscape (homogenization vs. increased heterogeneity), the type or combination of biomass crops planted and the potential invasiveness of the crops and genetic contamination of wild plants. At the regional scale, we compare prognoses for future land area demanded by biomass crops and respective effects of the crops across regions and for areas of different nature conservation value.

Field-scale studies on biomass crops and biodiversity

An intensive literature search provided us with a total of 47 publications and reports from nine European countries and the USA which studied the impacts of biomass crop production on biodiversity (Table 1 and Appendix S1). The majority of biomass crops studied were SRC crops in particular willow, poplar (*Populus* spp.) or mixed plantations, and the perennial grasses *Miscanthus* spp., reed canary grass (*Phalaris arundinacea*) and switchgrass (*Panicum virgatum*). Most of the European studies, both on SRC and perennial grasses, were conducted in the UK and Germany. The North American studies focussed primarily on willow, poplar and switchgrass.

It is likely that this list of studies is biased towards studies written in English and German because the authors were able to intensively search library databases and internet resources in those languages. Cross-references only rarely directed us to studies from non-English- or non-German-speaking countries. 43% of the studies we found were not published in international and/or easily accessible journals but were reports (3), conference proceedings (2) or publications in smaller, national journals (15). It is likely that several more European studies do exist, but results are hidden in reports or conference proceedings. Studies focussing on weed or pest control in energy crops were not included in the list as they provided very limited information about the biodiversity in the crops but the implications for biodiversity arising from crop management were included in the discussion of field-scale impacts.

Most of the studies on SRC crops focused on birds, both breeding and winter birds (Table 1). Some of the North American studies also included mammals and very few looked at soil fauna. Many European studies, both on SRC and perennial grass crops, focused on, or included in addition to birds, vegetation and various groups of canopy-, ground- and soil-living invertebrates (Table 1 and Appendix S1). The comprehensiveness of the studies varies considerably in terms of number of sites surveyed, age of sites and sampling strategy used (see also the section on 'Knowledge gaps at the field scale').

The vast majority of studies compared species richness, abundance and species composition in energy crop plantations to other types of land use, either of the surrounding landscape or land use replaced by the respective plantations (Table 2). The land use most frequently compared with both SRC and perennial grasses was arable land (in particular land used for winter wheat, maize, barley and oilseed rape) and the SRC plantations were compared with natural woodlands or managed forests. Few studies compared biomass crops with grassland, set-aside or noncultivated land.

The number of studies comparing different types of energy crops are also limited (Table 2). Three studies compare reed canary grass with either *Miscanthus* × *giganteus* (Semere & Slater, 2007a, b) or with other fast growing perennial grasses (Jahnova & Bohac, 2009). A Swiss study compared *Miscanthus sinensis* with hemp (*Cannabis* sp.) and kenaf (*Hibiscus cannabinus*) and a variety of other renewable primary products (Loeffel & Nentwig 1997). In the USA, willow was compared with poplar (Dhondt *et al*., 2004) and SRC sweetgum (*Liquidambar styraciflua*) with switchgrass (Ward & Ward, 2001). Several studies did not compare the respective energy crops with any other type of land use but instead compared age structure, plant height or different clones within the energy crops (e.g. Coates & Say, 1999; Berg, 2002; Dhondt *et al*., 2004; Boháč *et al*., 2007; Gruß & Schulz, 2008). Nevertheless, those studies provide valuable information about the importance of energy crop management, heterogeneity, canopy structure and harvesting patterns for various species or species groups (Appendix S1).

Effects of biomass crops on biodiversity at the field scale

Crop and land-use comparisons. Second generation biomass crops are perceived as being beneficial for

Table 1 Number of studies investigating diversity and/or composition of various groups of fauna and flora in biomass crops in temperate regions of Europe and the USA

		Number of studies									
Biomass crops	Species groups	Total	USA	UK + ROI	D	DK	S	CZ	PL	I	CH
Short-rotation coppice											
Willow	Birds	7	1	3		2	1				
	Mammals	1				1					
	Butterflies	3		3							
	Canopy invertebrates	2		2							
	Earthworms	3			2	1					
	Other soil fauna	1	1								
	Plants	5		4			1				
Poplar	Birds	3	2	1							
	Mammals	2	2								
	Canopy invertebrates	1		1							
	Ground beetles	5		1	1			1	1	1	
	Rove beetles	2		1				1			
	Spiders	2		1	1						
	Earthworms	2			2						
	Other soil fauna	1			1						
	Plants	4		2	1		1				
Willow/poplar mix	Birds	5	1	2	2						
	Mammals	1		1							
	Ground beetles	2		1	1						
	Rove beetles	1		1							
	Spiders	1		1							
	Ground invertebrates	1			1						
	Earthworms	1		1							
	Plants	2		1	1						
Sweetgum	Ground beetles	1	1								
Perennial grasses											
Miscanthus × giganteus	Birds	4		3	1						
	Mammals	3		2	1						
	Butterflies	2		2							
	Canopy invertebrates	2		2							
	Ground beetles	1		1							
	Spiders	1			1						
	Ground invertebrates	2		1	1						
	Earthworms	2		1	1						
	Plants	2		2							
M. sinensis	Ground beetles	1									1
Reed canary grass	Birds	2		2							
	Mammals	2		2							
	Butterflies	1		1							
	Canopy invertebrates	1		1							
	Ground beetles	2		1				1			
	Rove beetles	1						1			
	Plants	2		2							
Switchgrass	Birds	1	1								
	Ground beetles	1	1								

CH, Switzerland; CZ, Czech Republic; D, Germany; DK, Denmark; I, Italy; PL, Poland; S, Sweden; UK + ROI, United Kingdom and Republic of Ireland.

Table 2 Comparison of the intrinsic biodiversity value of biomass crops with a range of cultivated and uncultivated habitat types as well as with other biomass crops

		Arable land			Grassland			Woodland				Set aside/ degr.			Open uncultivated			Other Biomass		
Biomass crops	Species groups	H	L	E	H	L	E	H	L	E	C	H	L	E	H	L	E	H	L	E
Short-rotation coppice																				
Willow	Birds	4		1	1		1		2		2	2					1	1*		
	Mammals			1					1					1		1				
	Butterflies	3†																		
	Canopy invertebrates																	1*		
	Earthworms	3					1													
	Other soil fauna										1‡									
	Plants	2			1															
Poplar	Birds	2			1				2		1									
	Mammals			1					1	1										
	Canopy invertebrates																			
	Ground beetles		3					1		1										
	Rove beetles		1																	
	Spiders	2§				1														
	Earthworms	1					1													
	Plants	1		1																
Willow/poplar	Birds	1							1	1	4¶							1‖		
mix	Ground beetles		1					1												
	Ground invertebrates	1						1												
	Plants	1																		
Sweetgum	Ground beetles			1																
Grasses																				
Miscanthus × giganteus	Birds	2					1								1			1**		1**
	Mammals	1			1															2**
	Butterflies	1†																1**		
	Canopy invertebrates			1														1**		
	Ground beetles																	1**		
	Spiders	1													1					
	Ground invertebrates			2											1					
	Earthworms	1		1††																
	Plants	2																1**	1**	

Continued

M. sinensis	Ground beetles	1	1		1‡‡	
Reed canary grass	Birds			1		
	Mammals		1			
	Ground and rove beetles					1§§
	Plants	1				
Switchgrass	Ground beetles	1			1¶¶	

Presented is the number of studies reporting higher (H), lower (L) or equal (E) species richness in biomass crops. C, number of studies reporting differences in community composition between SRC crops and woodlands; degr., degraded land.

*In comparison with poplar plantations.

†Comparison of field margins and headlands.

‡Mite communities resemble early forest succession in more mature stands.

§Only in old plantation (9 years) not in young plantation (4 years) in the study by Blick & Burger (2002).

¶Number of woodland species increases with age of plantations.

||Breeding density higher in mixed plantations than in pure willow stand.

**In comparison with reed canary grass.

††Biomass and abundance.

‡‡In comparison with hemp and kenaf.

§§Compared with *Dactylis glomerata*.

¶¶In comparison with sweetgum.

biodiversity compared with cultivated areas of arable food crops because, in general, biomass crops have longer rotation periods, low fertilizer and pesticide requirements, provide better soil protection, a greater richness of spatial structures, are exposed to fewer disturbances during the growing period, and harvesting is carried out in winter or can be done after the breeding period of birds, which again causes less disturbance (EEA, 2007; Haughton *et al*., 2009; Rowe *et al*., 2009). Indeed, when compared with arable fields, all types of biomass crop plantations showed a positive effect on species richness for almost all taxa studied (Table 2). Only ground beetles (Coleoptera: Carabidae), and, in some of the studies, rove beetles (Coleoptera: Staphylinidae) were found to have higher species richness in arable land than in SRC crops (Liesebach & Mecke 2003; Ulrich *et al*., 2004; Britt *et al*., 2007). Weih *et al*. (2003) reported a similar number of plant species across all sites of poplar stands and arable fields but a small number of species were shared between both types of land use.

In contrast, most studies comparing SRC crops and woodland habitats reported lower species richness in the energy crops or no significant differences. Positive effects were only recorded for ground beetles and other ground living invertebrates. The most important findings of these studies, in particular when focussing on birds, was that the species composition of the SRC crops did not resemble forest bird communities but open farmland or transitional scrubland communities (Hanowski *et al*., 1997; Liesebach & Mulsow, 2003; Reddersen & Petersen, 2004). Plots planted with fast growing trees are extremely dynamic and within 4 years they can change from being open habitat to being young forest-like habitat, with trees reaching 10–15 m in height. Consequently, with increasing age of the plantations, forest elements of the bird fauna become more common in the SRC plots (Göransson, 1994; Gruß & Schulz, 2008; Kroiher *et al*., 2008). In the context of pastoral landscapes, SRC crops create a pseudo-arable environment for weeds during the establishment phase but create early succession woodland conditions when mature (Fry & Slater, 2008; Valentine *et al*., 2009). Accordingly, the number of woodland indicator plant species increases in poplar plantations compared with agricultural surroundings (Britt *et al*., 2007). When compared with forests, SRC plantations may contain higher plant species richness than coniferous forests, but poorer than in old-growth mixed deciduous forests (Baum *et al*., 2009).

Compared with uncultivated land, the few studies which included set-aside land reported higher bird species richness in SRC willow plantations but no differences for small mammals (Reddersen *et al*., 2001;

Reddersen & Petersen, 2004; Reddersen *et al.*, 2005). Similarly, higher species richness was found for birds, spiders and ground invertebrates in *Miscanthus* × *giganteus* compared with uncultivated stands of *Phragmites australis* (Jodl *et al.* 1998, 2004), but no differences were found in bird species richness between SRC willow and uncultivated fens (Reddersen & Petersen, 2004). Mammals had lower species richness in SRC willow than in small biotopes such as shelterbelts, grassy ditches or canal banks (Reddersen *et al.*, 2005).

Findings reported from the few studies that compared species richness of biomass crops with grasslands were ambiguous. For woody crops, no differences were found in earthworms in SRC willow compared with grasslands (Tischer *et al.*, 2006); willow and poplar had either positive or no effects on birds (Christian *et al.*, 1997; Reddersen & Petersen, 2004); and fewer spider families were recorded in a poplar plantation compared with a ley (an arable field being temporarily used as a pasture) (Britt *et al.*, 2007). Plant species richness and proportions of annuals and short-lived perennials was higher in recently planted SRC willow fields compared with grasslands, but declined again with increasing age of the plantations (Fry & Slater, 2008). For perennial grass crops, the abundance of individual birds was higher in *Miscanthus* × *giganteus* than in unimproved grassland but there was no difference in the number of bird species (Clapham & Slater, 2008); and more ground beetle species were found in *M. sinensis* than in managed grassland (Loeffel & Nentwig, 1997).

The comparisons among different biomass crops showed higher species richness of birds and canopy invertebrates in SRC willow than in SRC poplar plantations (Sage & Robertson, 1996) but breeding bird density was higher in mixed plantations than in pure willow stands (Kavanagh, 1990). This indicates that the physical structure of the vegetation is an important factor (Sage & Robertson, 1996; Schulz *et al.*, 2009). Higher species richness was reported for birds, plants and several invertebrate groups in *Miscanthus* × *giganteus* than in reed canary grass (Semere & Slater, 2007a, b), but differences between the crops for birds and mammals were less pronounced and weed species richness was even higher in reed canary grass than in *Miscanthus* × *giganteus* in the study of Clapham & Slater (2008). Species richness of ground beetles was higher in *Dactylis glomerata* grassland than in reed canary grass fields (Jahnova & Bohac, 2009).

Biomass crop management, age and heterogeneity. In general, positive effects of biomass crop cultivation on biodiversity are expected in the long term mainly due to the reduced soil tillage and use of agrochemicals and to the increased input of litter (Börjesson, 1999; Smeets *et al.*, 2009). However, many studies have reported considerable differences in biodiversity among biomass crop plantations, mainly due to differences in weed control measures, vegetation structure and heterogeneity and harvesting patterns (e.g. Göransson, 1994; Hanowski *et al.*, 1997; Dhondt *et al.*, 2004; Minor *et al.*, 2004).

The development of ground vegetation within the plantations is of great importance for the richness of the associated invertebrate communities and for the food availability and shelter for birds (Ward & Ward, 2001; Sage *et al.*, 2006; Semere & Slater, 2007a, b; Fry & Slater, 2008; Bellamy *et al.*, 2009; Valentine *et al.*, 2009). As commercial biomass crops are developed as monocultures which are managed mainly for maximum yield rather than biodiversity, weed control to avoid yield suppression and secure crop establishment is essential for SRC crops during establishment (Clay & Dixon, 1995) and for perennial grass crops such as *Miscanthus* during establishment and annual regrowth (Bullard *et al.*, 1995; see also Baum *et al.*, 2009 for a review on site preparation). As weed problems become less severe with maturation of the crops (Bullard *et al.*, 1995; Baum *et al.*, 2009), a succession or introduction of a stable perennial ground flora in biomass crops may be desirable, both from an ecological and economical viewpoint (Sage, 1999), and mechanical treatments for weed control may be preferable (Baum *et al.*, 2009).

SRC plantations support a diverse community of invertebrate species (Sage & Tucker, 1997). However, many of those species are pest species such as leaf-eating beetles (Coleoptera: Chrysomelidae) which can reach damaging numbers in the plantations (Sage, 2008). In plantations of perennial grass crops, which often are nonnative species, impacts of invertebrate pests are less severe. To prevent yield losses, pest control in SRC plantations could become necessary which would also suppress the nonpest invertebrates. Considering the negative effects on biodiversity but also the small economic profit margins of biomass crop cultivation, application of insecticides might not be a sensible option in SRC crops (Björkman *et al.*, 2004). As the leaf-eating beetles colonize the crops from the edge in each year, insecticide application in the edges only and during beetle colonization could be an effective, cost efficient and more biodiversity friendly method of pest control (Sage, 2008). Björkman *et al.* (2004) state that biological control is the only realistic way for pest management in SRC crops. Yet, biological control is disrupted by harvesting in winter as generalist predators, in contrast to many herbivores, overwinter within the crops and take longer to recover from the disturbance than the pest species do. Therefore, in order to support the biocontrol agents, a longer period between harvests that enables them to fully respond numerically and/or to disperse from refuges during harvest, or harvesting neighbouring plantations asynchronously would be necessary management strategies (Björkman *et al*., 2004).

Harvesting times and patterns have been reported to be of importance for other species groups as well. Harvested fields of switchgrass support a different grassland bird community than nonharvested fields due to a higher and denser vegetation structure (Roth *et al*., 2005). Harvesting of some fields and leaving other fields unharvested would increase the heterogeneity of vegetation structure and hence the overall bird diversity (Roth *et al*., 2005). Also in SRC crops, different species prefer different growth stages and a succession from harvested to mature crops can be observed (Göransson, 1994; Kroiher *et al*., 2008). In general, high-structural complexity enhances the species richness and abundance of birds within plantations (Nájera & Simonetti, 2009). Recommendations regarding the length of periods between harvests are conflicting and depend on the species groups in question. As many bird species prefer tall willow plants for nesting, relatively long periods between harvests would be beneficial for many bird species (Berg, 2002), whereas short periods would be beneficial from a plant conservation point of view because of the reduced light intensity in older stands (Gustafsson, 1987). In *Miscanthus* plantations, benefits for biodiversity were found in the first five years during crop establishment but declined when the crop became denser (Bellamy *et al*., 2009). Altogether, irregular harvest and/or establishment patterns of both SRC and grass crops would increase habitat diversity and species turnover and therefore the overall biodiversity. In addition, planting more than one clone and, if possible, species in a SRC stand would also increase the vegetation heterogeneity and hence the diversity of structural niches for a variety of species (Gustafsson, 1987; Dhondt *et al*., 2004; Londo *et al*., 2005), as well as increase the resistance of the plantations to leaf eating insects and diseases such as rust (Peacock *et al*., 1999; McCracken *et al*., 2005). Planting of willow species and clones with varying flowering times would extend the flowering season and may provide a more continuous resource for flower visiting insects (Reddersen, 2001). Furthermore, planting both male and female plants provides a more diverse SRC willow biotope, as males will produce both nectar and pollen while females produce nectar and eventually seeds (Reddersen, 2001).

In SRC and *Miscanthus* plantations, headlands and rides provide access to crops for harvesting and other management operations. Those small noncrop areas in the fields are an important feature for biodiversity. In intensively managed farmland they present an opportunity for rough grassland and wood-edge communities to exist (Sage, 1998). In particular, butterfly abundance and occurrence have been shown to be positively influenced by biomass crop headlands and margins (Sage, 1998; Anderson *et al*., 2004; Haughton *et al*., 2009). Among the butterflies, numbers of browns (Lepidoptera, Satyrinae) and skippers (Lepidoptera, Hesperiidae), both of which are associated with specific larval food plants, were higher in SRC headlands than in arable headlands and those two groups were also positively affected by increasing width of the headlands (Sage *et al*., 2008). Edges of hybrid poplar plantations located in agricultural land showed strong edge effects for butterflies with high numbers of individuals found in the plantations resembling open woodland edges (Britt *et al*., 2007). The number of plant species, in particular of perennials, was greater in SRC headlands than in arable headlands (Sage *et al*., 2008). An active management of headlands to maximize plant species richness, a diverse vegetation structure and floral richness could support a species-rich invertebrate community providing ecosystem services such as pollination and biocontrol for the biomass crops themselves and/or the arable crops in the vicinity (Sage, 1998; Sage *et al*., 2008; Bellamy *et al*., 2009; see also Marshall *et al*., 2006).

Plantation size and edge effects. The absolute size of the stand can affect the number of taxa present in biomass plantations, for example, plant species richness increased with increasing size of a SRC plantation but only until a size of 0.1–0.3 ha was reached (Kroiher *et al*., 2008). In addition, the shape of a stand and the relative edge-to-stand area relationship can affect biodiversity. Plant species richness decreases from the edges towards the central parts of poplar stands (Weih *et al*., 2003) and the possibility of plant propagules entering a SRC plantation does to some extent depend on the shape of the stand, with long, narrow stands having longer edges into which seeds could enter via wind or animal dispersal (Gustafsson, 1987). Also, species diversity of small mammals and birds was found to be higher in the borders of *Miscanthus* fields compared with the centres (Semere & Slater, 2007a). On large poplar plantations, lower overall bird densities were observed in plantation interiors than on edges (Christian *et al*., 1998). In SRC plantations, an edge effect on birds became apparent with increased time since last harvest (Cunningham *et al*., 2004). The edges of the SRC crops contained higher bird abundance and the hedgerow boundaries contained higher bird abundance and higher diversity than hedgerows around arable crops.

Multifunctional land use: waste and wastewater application. The rapid growth and relatively frequent harvests of SRC crops remove nutrients from the soil. Therefore, to maintain soil productivity, nutrients have to be artificially applied in similar quantities to those removed during harvest. Because these crops are not for human consumption, waste organic material in form of municipal wastewater, landfill leachate and sewage sludge, can be applied to the plantations instead of industrial fertilizer. SRC crops such as willow and poplar transpire large volumes of water, have an extensive root system and a long growing season which makes them ideal candidates for irrigated vegetation filter systems (Berndes *et al*., 2008). A multifunctional land use of biomass production and waste disposal secures the return of nutrients into the soil, irrigates the plantations and further enhances the economic feasibility of the system (Abrahamson *et al*., 1998). Although the application of waste materials to biomass crops, such as SRC and *Miscanthus*, might be expected to have significant effects on the flora and fauna, an extensive literature review by Britt (2002) revealed almost no evidence of research that looked directly at the ecological effects of applying farm, urban or industrial waste products to biomass crops. Thick applications of waste materials may have a net detrimental 'mulching' effect and suppress the ground flora, whilst organic wastes may provide a valuable additional food source for soil and ground-dwelling microorganisms and invertebrates with a generally positive effect and both would have 'knock-on' effects up through the food chain (Britt, 2002). The bioaccumulation of heavy metals, organic toxins, polycyclic aromatic hydrocarbons in animal tissues and increased exposure risk to pathogens are of important concern but generally not well studied within the biomass crop context (Bain *et al*., 1999; Britt, 2002; Minor & Norton, 2004).

In some countries (e.g. Sweden), the willow vegetation filter system has become established on a large scale and its potential for phytoremediation is mostly regarded as beneficial, from a waste management point of view (Berndes *et al*., 2008). From a biodiversity and sustainability point of view, concerns regarding the generalization of beneficial effects of the vegetation filter systems are: (i) the nutrient composition in waste products is often different to the requirements of the plants and (ii) the removal of nutrients from the system by harvests is limited. To avoid leaching of nutrients, amounts and composition of wastewater should be regulated accordingly (Nielsen, 1994).

Persistent organic contaminants may transfer through the food chains and cause adverse effects on human health or on soil fauna and flora after long-term application (Chen *et al*., 2005). Bioactive substances such as antibiotics, as well as resistant microorganisms from contaminated excrements, can cause resistance in soil microorganisms, directly or via gene transfer. This increases the risk of infections in humans and animals that can not be treated with pharmaceuticals (Thiele-Bruhn, 2003; Carlander *et al*., 2009). Anthelmintics were reported to increase mortality in springtails and inhibit nematodes and earthworms in soil (Thiele-Bruhn, 2003).

Knowledge gaps at the field scale. As the compilation of studies and observations shows (Tables 1 and 2), negative, neutral and positive impacts of biomass crops on biodiversity at the field scale exist and the direction of the impact strongly depends on the respective crop, the land use replaced, the landscapes and biogeographical context in which the plantations are embedded and of course on the group of organisms considered.

Given the scale of biomass production anticipated within temperate regions of Europe and North America (Graham, 2007; Hellmann & Verburg, 2008), the number of published studies on potential effects on biodiversity is very small. What is further limiting our knowledge about effects on biodiversity and the ability to come to a general conclusion is that the number of studies with comprehensive and scientifically sound study designs is limited; 14 out of 36 studies on SRC crops and eight out of 11 studies on perennial grass crops were based on fewer than five replicate sites and lacked proper controls in some cases (Appendix S1). Therefore, many of the findings are based on singular observations and are difficult to generalize. In many countries only a few commercially used fields of biomass crops exist. In consequence, about half of the existing studies were conducted on experimental SRC or grass plots and the inferences that can be drawn for extensive commercial production of these crops are limited as different pictures might emerge for full-commercial SRC production (Anderson *et al*., 2004). Fortunately, the importance of more thorough and interdisciplinary projects on environmental impacts of biomass crops has recently been recognized by national and international funding bodies and outputs from the established projects can be expected in the coming years.

Since the age and maturity of the energy crop is of great importance for the patterns of species richness found within the crop (Bellamy *et al.*, 2009; Schulz *et al.*, 2009), crop age needs to be considered within study designs. However, for the perennial grass crops in particular, only a few studies were able to include old plantations as production in many countries only started a few years ago and no mature crops were available for the studies. Ecological studies are further constrained by the limited spectrum of ages, the small size of most available plantations, poor establishment of some younger plantations, the lack of statistical independence between age and plantation size, and by a limited replication of landscape contexts (Christian *et al.*, 1998; Rowe *et al.*, 2009). Caution is advised for extrapolating conclusions about habitat or biodiversity value of biomass plantations from one landscape to another particularly as there is a tendency for different responses in forested as opposed to agricultural regions (Christian *et al.*, 1997). Contradictory results may also arise from comparisons of biomass plantations located in different bio-geographical regions. For example, in the American North-West, large plantations of SRC poplar were found to decrease floristic diversity compared with old-growth forest, whereas small-scale SRC willow cultures in Sweden were found to increase biodiversity compared with coniferous forests (Weih *et al.*, 2003).

An assessment of the impacts on biodiversity is further hampered by the fact that it is currently not clear which types of land use and habitats would be replaced by full-scale commercial production of energy crops. Only a few studies analysing the implications of land allocation to biomass cropping which incorporated yield variations and other land-use characteristics exist (Lovett *et al.*, 2009). As any change of land use will affect some species positively and others negatively, it is important to identify the priorities for biodiversity conservation with respect to expected landscape change (Firbank, 2008).

The identification of useful biodiversity indicators for agricultural landscapes is currently an important topic in biodiversity research (Büchs, 2003; Billeter *et al.*, 2008). Findings about the usefulness of certain species groups as indicators are; however, contradictory and the concept of using indicator taxa has been questioned in general (Wolters *et al.*, 2006; Wugt Larsen *et al.*, 2009). Therefore, a more thorough consideration about the biodiversity indicators surveyed in biomass crops should perhaps be a priority for future studies. Haughton *et al.* (2009) have suggested butterflies as an appropriate ecological indicator for biomass crops but this would need further confirmation for widespread use as their study was confined to field margins of biomass crops in England.

Interactions between energy crops and surrounding landscapes

Species interchanges between energy crops and surrounding land use. The importance of size and shape of plantations for biodiversity at the field scale has been discussed above. Looking beyond the field scale, size and shape of plantations are important factors for the interactions between biomass crops and other land use in the vicinity. For example, in small SRC plantations, a high portion of animals moved from adjacent habitats into the plantations (Christian *et al.*, 1997; Hanowski *et al.*, 1997) although colonization of poplar plantations by woodland carabids was only observed when dispersal from surrounding woodlands is possible (Allegro & Sciaky, 2003). Also, a high number of arthropods from surrounding arable crops, among them potential biocontrol agents, were found overwintering in *M. sinensis* plantations (Loeffel & Nentwig, 1997) and locating perennial grass fields of optimal size close to different types of vegetation increases their biodiversity (Smeets *et al.*, 2009). These examples indicate that species communities on plantations are influenced fundamentally by the surrounding landscape (e.g. Christian *et al.*, 1998). In turn, as biomass crops might act as temporal habitat or shelter for species, the plantations could also have a positive feedback effect on landscape scale biodiversity and on ecosystem service performance in neighbouring habitats (e.g. EEA, 2007; Porter *et al.*, 2009). An optimization of biomass crop field sizes with a larger number of smaller plantations interspersed in the landscape may therefore be desirable from a biodiversity and ecosystem service perspective (Gustafsson, 1987; Wissinger, 1997; Perttu, 1998; Smeets *et al.*, 2009). This perspective; however, is likely to resurrect the SLOSS debate (are single large areas better for biodiversity conservation than several small areas?) by introducing biomass crops into the equation.

Invasiveness and genetic pollution. Currently, perennial rhizomatous grasses are among the leading candidates for biomass energy production. They are selected and bred for desirable agronomic traits such as tolerance to drought and low soil fertility, as well as high aboveground biomass and enhanced competitive ability against weeds, making a reduction in fertilizer and pesticide applications possible (Barney & diTomaso, 2008). Those plant traits that characterize an ideal energy crop; however, also contribute to a higher probability of naturalization and invasiveness (Barney & diTomaso, 2008; Buddenhagen *et al.*, 2009). Indeed, some of the most promising biomass crops are nonnative to Europe and North America, hence holding the potential risk of future invasions. By applying a weed risk assessment system for screening out potentially invasive species, Buddenhagen *et al*. (2009) found that energy crops, in comparison with nonenergy crops, were two to four times more likely to be naturalized or invasive. Subjecting switchgrass, giant reed (*Arundo donax*), and *M*. spp. to a weed risk assessment protocol, which took biogeography, history, biology, and ecology into account, revealed switchgrass to have a high invasive potential in California, giant reed to have a high invasive potential in Florida, and *Miscanthus* × *giganteus* (a sterile hybrid) to pose little threat of escape in the USA (Barney & diTomaso, 2008). Some small-scale escapes of fertile ornamental *Miscanthus* genotypes have been reported from Ohio and Indiana, USA, and therefore it was recommended that new *Miscanthus* genotypes should be sterile (e.g. triploid) hybrids as a precaution against them becoming invasive weeds (Lewandowski *et al*., 2000). As there has been little success so far in eradicating or even controlling invading grasses, the ecological risks of introducing biomass crops must be rigorously assessed before starting their cultivation (Raghu *et al*., 2006). Barney & diTomaso (2008) suggest a genotype-specific preintroduction screening for a target region, consisting of risk analysis, climate matching modelling, and ecological studies of fitness responses to various environmental scenarios. The risk of invasiveness also needs to be addressed when, after the introduction of the species, land users are planning the location of the energy crop in a landscape. It would for example be inappropriate to plant giant reed adjacent to high-quality wetlands as it is known to invade riparian habitats (Firbank, 2008).

Genetic contamination of wild native species through hybridization with nonnative crop species is another risk associated with large-scale introduction of biomass crop cultivation (Firbank, 2008). Solutions to this problem could be the development of sterile clones of nonnative species or using female clones only, or the use of native species as it is practised with willow species in Sweden (Börjesson, 1999). This would minimize the movement of genes from crops into native gene pools and avoid genetic pollution of native taxa. The impact of gene transfer to wild relatives is considered an important risk to biodiversity should the crop be genetically modified or should the crop be located in an area of genetic diversity (Firbank, 2008).

Landscape heterogeneity and spatio-temporal habitat mosaics

At the landscape scale, plantations of biomass crops may influence spatial and temporal ecological processes taking place across field boundaries, altering species interactions and responses of populations and communities (Christian *et al*., 1998; Firbank, 2008). Biomass crops could destroy seminatural habitats but they could also act as buffer areas, shelters or ecotones, hence they could increase or decrease habitat fragmentation and availability. Understanding impacts of land-use change on biodiversity requires developing information and perspectives at broader spatial scales than the plantations themselves (Christian *et al*., 1998; Tscharntke *et al*., 2005). This means that for an impact assessment, the mainly positive effects of biomass crop plantations on species richness, reported by the majority of surveys undertaken at the scale of individual fields (Table 2), would require an upscaling by taking landscape scale ecological processes into account.

Biomass crop production has the potential to change the diversity of land use making it either more uniform in the case of extensive, large-scale monocultures (see section on ‘Regional effects’) or more diverse in the case of smaller polyculture plantations interspersed in a previously homogeneous landscape (Firbank, 2008; Williams *et al*., 2009). Its impact also depends on whether the landscape is characterized by annual or perennial cropping systems. Biomass crops are likely to increase biodiversity in areas dominated by agriculture or coniferous forests but they are not likely to provide wildlife habitats of major significance because the plantations rarely harbour species that would not be found elsewhere in the surrounding landscape (Britt *et al*., 2007; Baum *et al*., 2009; Schulz *et al*., 2009). In consequence, biomass plantations could potentially have adverse effects in landscapes of high conservation value (Anderson & Fergusson, 2006; Schulz *et al*., 2009).

Habitat heterogeneity at a range of spatial scales has been identified as one key issue for maintenance of farmland biodiversity (Benton *et al*., 2003). The introduction of biomass crops in farmed landscapes could improve habitat heterogeneity, thereby preserving natural biodiversity and simultaneously diversifying the income mix of landowners (Cook & Beyea, 2000), possibly sustaining farming in marginal high nature value farmlands (cf. Bignal, 1998) or helping to restore degraded land (UNEP, 2009). However, whether production of biomass crops on marginal or degraded land would be possible and economically sustainable is questionable (Howarth *et al*., 2009). If productivity is lower, then a larger land area would be required to meet specific targets, and it is more likely there will be conflicts with other landscape functions and ecosystem services (Berndes *et al*., 2003; Huston & Marland, 2003).

Currently, we have limited understanding of the proportion of land covered by biomass crops that would significantly affect species richness or survival of wildlife populations in landscapes. A proportion of 10–20% SRC crops in open farmland has been estimated to be optimal for bird species number (Göransson, 1994). In general, it has been suggested that a threshold of 20% represents a minimum amount of habitat needed in a landscape, below which effects on population persistence become evident (Fahrig, 1998). For example, it has been shown that proportion of parasitism of rape pollen beetles drops below a level necessary for efficient biocontrol of the beetle when the percentage of noncrop area drops below 20% (Tscharntke *et al.*, 2002). Observations of limiting thresholds of habitat cover exist for various species groups (e.g. Garaffa *et al.*, 2009; Utz *et al.*, 2009). Therefore, future studies should verify whether such thresholds apply for perennial grass plantations or SRC cropping systems.

Habitat heterogeneity of a landscape, in the light of biomass crop cultivation, depends not only on the proportion and the diversity of the crops but also on the turnover of the biomass crop fields. Owing to the long rotation periods of 20 plus years, the 2–3 year establishment phases of perennial grasses during which the fields are patchy in terms of crop development and the 3–5 year harvest cycles of SRC crops, biomass crops provide transitional habitats which vary considerably in vegetation structure and habitat quality. When planting and harvesting is done asynchronously on a landscape-wide rotating routine, the resulting spatio-temporal mosaic of sites could provide high structural diversity for a variety of species (Sage & Robertson, 1996; Tolbert & Wright, 1998; Börjesson, 1999; Smeets *et al.*, 2009). An irregular harvest pattern of switchgrass fields, for example, provides habitat for a larger number of bird species than if all fields are harvested simultaneously (Roth *et al.*, 2005). Large-scale SRC plantations containing fields of different age classes and a variety of crop species or clones support a more diverse community of species (Sage & Robertson, 1996; Dhondt *et al.*, 2007; Baum *et al.*, 2009). Furthermore, cultivating a variety of energy feedstock could increase habitat diversity in agricultural landscapes and enhance arthropod-mediated ecosystem services (Landis *et al.*, 2008).

Integration of biomass crops into existing landscapes

A strategic landscape design and judicious positioning of plantations, especially in homogeneous annual crop-dominated landscapes, could result in overall positive effects on biodiversity. At this point, biomass crop cultivation could provide the opportunity to build some ecological and ecosystem service values into the existing land-use systems which have not traditionally been considered (Paine *et al.*, 1996).

Biomass crops, in particular SRC crops with high vegetation filter potential, could be planted as buffer strips along streams or other (semi-) aquatic habitats to maintain good water quality and hence aquatic biodiversity in agriculturally dominated landscapes (Abrahamson *et al.*, 1998; Börjesson, 1999). As about 70% of the water's nitrogen content is estimated to be removable by willow strips of 25–50 m width, a 50 m wide buffer, where half of the width is harvested at a time, could provide a continuous high uptake of nutrients (Berndes *et al.*, 2008). This service of water purification could; however, compete with the conservation value of some riparian sites which is linked with the openness of the habitat (Berg, 2002).

SRC crops could also be planted along sharp edges between coniferous plantations and open farmland where they function as ecotones in order to increase the complexity of the forest margins (Berg, 2002). In contrast, plantations in forest-dominated landscapes could have negative effects, since the mosaic structure of open and forest habitats, which is positive for most farmland birds, would disappear, and only a few forest bird species are favoured by SRC plantations (Berg, 2002). To improve quality of remnants of natural habitats for native wildlife, perennial biomass crops could also be integrated with annual crops as buffers around natural areas or woody crops around forest remnants (Cook & Beyea, 2000). Suggestions that SRC plantations might enhance connectivity of forest habitats should be treated with caution because SRC plantations do not represent forest habitat for many taxa and better evidence of their use as dispersal corridors would be needed (Christian *et al.*, 1997). Porter *et al.* (2009) suggest using biomass belts, i.e. 11 m wide rows of clonally mixed fast-growing bush willows (*Salix* spp.) with two rows of alder trees (*Alnus rubra* which fixes N) on one side and two rows of hazel bushes (*Corylus* spp. which are attractive to predatory insects) on the other side, to increase ecosystem services such as biological pest control in arable crop/pasture systems. The biodiversity value of such biomass belts would have to be compared with the value of traditional hedgerows if they were to replace them. Such mapping exercises of biomass crop positioning are however only realistic if the demand from biomass crops for land area would allow an integration of biomass crops into landscapes and not dictate an overall conversion of land to biomass crops as discussed in the following section on regional effects.

Regional scale of biomass crop development

The anticipated large increase in biomass crop production will require large land areas (UNEP, 2009). As high-quality land for agricultural production is limited, energy crops will most likely compete with food and feed production, urban development, forestry and nature conservation (Sala *et al*., 2009). The potential of future biomass production and distribution has been calculated based on various combinations of feedstock and materials, socio-economic and land-cover scenarios and future climate conditions resulting in a wide range of estimates (Berndes *et al*., 2003; van Dam *et al*., 2007; Bustamante *et al*., 2009). No matter which estimate of the extent of biomass crop development is correct, even meeting the lowest targets would result in major changes of land use (Hellmann & Verburg, 2010).

According to modelling approaches conducted at the scale of European biogeographical regions and based on land-use sensitivity of a set of 754 species, comprising mammals, reptiles, amphibians, birds, butterflies and vascular plants (Eggers *et al*., 2009; Louette *et al*., 2010), large-scale woody biomass crop production may have a negative net effect on the species considered (Louette *et al*., 2010). In this analysis, the individual species groups did show considerable differences in their response to the assumed land-use change, with negative effects being strongest for reptiles, butterflies and birds, whereas plant species would profit from the biomass crop production (Louette *et al*., 2010). These results have to be interpreted with caution because the generalized and coarse-scale approach chosen can only report relatively broad impact patterns. The major impacts and pressures of biomass-induced land-use change will however operate on the finer spatial scales of landscapes or regions and therefore such modelling approaches would have to be down-scaled in order to become more accurate by taking patterns of spatial distribution of biomass plantations and habitat replacement in the regions and the associated ecological processes into account.

When utilizing biomass feedstock for cofiring for example, locations of existing power plants, distance to markets or heat generating facilities, minimization of transport distances to maximize the economic benefit of the crop and the net reductions of GHG emissions, create economic pressures for processing or combustion plants to be surrounded by large areas of biomass feedstock (RCEP, 2004; Anderson & Fergusson, 2006). This could result in a spatial aggregation of energy crop plantations in regions providing appropriate infrastructural conditions (Hellmann & Verburg, 2010).

The distances for economic collection of biomass crops range from 45 to 90 km for SRC willow and 30 km for *Miscanthus* (RCEP, 2004). Consequently, this is reflected in the UK grant funding for biomass crop cultivation, which specifies that the crops should be grown within 40 km of the end user (Wildlife & Countryside Link, 2007). Examples for such aggregation processes are 1300 ha of SRC established in a 75 km radius of a biomass gasification plant close to York in the UK (Pitcher & Everard 2001) and 16 000 ha of willow SRC in the vicinity of four coal-fired power plants in the state of New York in the USA (Dhondt *et al*., 2007). According to Graham *et al*. (1996), an efficient biomass power plant would require between 200 and 400 ha of biomass crops (depending on the yield) per MW of baseload power. A 2000 MW power station cofiring 25% of its fuel from dedicated energy crops, at current yields of SRC willow in the United Kingdom, would require the willow biomass from ca. 1500 ha of land (RCEP, 2004). Given an approximate farm size of 150 ha for mixed farms in the United Kingdom, this would equal a total of 10 farms in the vicinity of the power plant to convert all their land to SRC plantations. In Finland, farmers have entered into contracts with power plants to grow reed canary grass for cofiring. A survey of 74 farms (i.e. 47% of all contracted farms) showed that on average an area of 14 ha, being more than one-third of the fields on a farm, were planted with reed canary grass and maximum transport distance of the crop was 80 km (Pahkala *et al*., 2008).

These trends of centralized biomass crop cultivation could result in large-scale monocultures in the most suitable locations and a segregation of landscapes for energy production from landscapes for food production and landscapes for nature conservation. Given such a scenario, ideas for strategic landscape design, using biomass crops as elements to increase structural diversity (Bellamy *et al*., 2009), ecotones or buffer zones to improve landscape quality for wildlife would be futile. Whether biomass crops have significant potential to be integrated into multifunctional landscapes that simultaneously advance production, conservation and livelihood goals (Berndes *et al*., 2008; Porter *et al*., 2009) depends on the markets and the structure of the infrastructure involved. One way to integrate biomass crops into agricultural landscapes is in the context of smallholder production for local use (Milder *et al*. 2008) and the development of local markets (Anderson *et al*., 2004). Efforts are underway at the US Department of Agriculture to evaluate biomass crops as an alternative to Conservation Reserve Program (CRP) set-asides (Cook & Beyea, 2000) and it might be worthwhile taking them into consideration in the framework of agri-environment schemes in the EU as well to encourage smaller plantings, the splitting of blocks by rides and hedges, and rotational harvesting in mixed age-class blocks (Sage *et al*., 2006). A scenario of land segregation between intensive food production, intensive biomass crop production, urban development and nature conservation, reduced to whatever land is left, would be a worst case situation for biodiversity. There is room for a rational and efficient use of biomass at the rural level and research into prospects of biomass production should be based on holistic and multiscale approaches.

Strategies for biodiversity friendly energy crop production

With policy-makers deciding on targets to increase the production and use of biomass resources before sound scientific knowledge about the risks of bioenergy production are understood (Florin & Bunting, 2009), there is an urgent need for recommendations on sustainable and biodiversity friendly production of biomass crops which range from the field via the landscape and regional scale to national and international policy level.

Recommendations for the field scale. Based on the knowledge gained from studies undertaken at the field scale, several publications list recommendations for biodiversity friendly and sustainable bioenergy production (e.g. Sage, 1998; Firbank, 2008; Groom *et al.*, 2008; Baum *et al.*, 2009; Rowe *et al.*, 2009). General guidelines for biomass production addressing in particular farm- and field-scale management are:

- Grow energy crops that require low fertilizer, pesticide, and energy inputs in most settings (i.e. biomass crops such as SRC crops or perennial grasses).
- Minimize pesticide impacts on the nontarget invertebrate population and promote biocontrol agents.
- Grow native species or varieties.
- Promote polyculture and/or use a mix of varieties (preferably of different gender and different growth structure) to increase within crop heterogeneity.
- Use willow clones with a range of flowering times to promote resource availability for flower visiting insects.
- Design plantations with large edge to interior ratio and incorporate rides and headland of at least 6 m in width.
- Introduce nectar sources into rides and headlands for flower-visiting insects.
- Intersperse blocks of biomass crops with other farmed habitats and keep plots' size below 15 ha.
- In large plantations, apply varying harvest cycles to individual plots and establish plots in different years to diversify the age structure (mixed-age stands).
- Encourage growth of weed species in crops after the establishment phase of the crop.
- Application of wastewater or other waste materials to the crops must comply with the nutrient uptake of the crop variety, the soil and hydrological situation of the site.

Efforts of farmers for biodiversity friendly production of biomass crops have to be supported by policy makers, landscape planners and by further research. A strategic regional specific landscape planning would be desirable to locate biomass plantations in such a way that they maximize variation in habitat type and can function as buffer, corridor or stepping stone habitat.

This would have to go hand in hand with research on plantation–landscape interactions. Furthermore, research needs to investigate production methods that may enhance biodiversity and ecosystem services over time. Also impacts of secondary land-use of biomass crops such as waste application would require a better knowledge-base so that specific local management recommendations could be provided.

Developing strategies for the landscape and regional scale. A matter of particular concern is the likelihood of large-scale land-use change induced by bioenergy production. A general recommendation is therefore to select feedstock with high conversion efficiencies to minimize land area needed to produce biofuels and only produce feedstock that are proven to be net carbon neutral or that sequester carbon (Groom *et al.*, 2008). As stated before, the primary target of bioenergy production is, or should be, climate change mitigation. From a biodiversity conservation perspective, the reason behind this is that climate change is expected to drive a large number of species into extinction (Pimm, 2008). The dilemma that biodiversity conservation is facing with bioenergy production is that either species could be lost due to climate change mitigation actions now, if biodiversity friendly cultivation of energy crops is not applied or is not possible, or species are lost later due to climate change if the mitigation was not successful because production necessary to reduce GHG emissions was not achieved. It is therefore important for biodiversity researchers to work closely together with other disciplines and decision makers to turn a potential lose–lose into a win–win situation. Given the importance of climate change mitigation, but also the extent of the impacts of bioenergy production, society can neither afford nor accept impacts of bioenergy cultivation gone wrong (Tilman *et al.*, 2009). The challenge for scientists is to investigate the potentials and risks of bioenergy and for decision makers to give consideration to recommendations in order to implement policies and encourage developments that ensure the full potential of bioenergy is realized without causing the associated risks to occur (Florin & Bunting, 2009).

In contrast to the field scale, no clear-cut recommendations for coarse scale management of biomass cultivation exist. As the perspectives of biomass production are varied across regions of Europe and North America, it will not be possible to apply a 'one size fits all' approach. In intensive agricultural regions for instance, introduction of biomass crops will most likely not impose additional pressures on biodiversity (Biemans *et al*., 2008), but might even improve their state. In high nature value farming systems, which are characterized by low intensity farming and high abundance of abandoned land (EEA, 2004), biomass production could in contrast pose a significant threat to biodiversity (Biemans *et al*., 2008; Hellmann & Verburg, 2010). Nevertheless, production of biomass crops or use of biomass from existing low-input grasslands could also potentially stimulate the economic profitability of farming within marginal regions and hence benefit biodiversity by counteracting abandonment of farming and loss of high nature value habitats (e.g. Hennenberg *et al*., 2009). Restoration of highly degraded land through production of biomass crops (Groom *et al*., 2008) as well as paying landowners to maintain environmentally sensitive land out of row crop production and under permanent grass cover and harvesting biomass from those lands might provide environmental benefits (Paine *et al*., 1996). Biodiversity benefits will only arise as long as sustainability principles are applied and novel farming activities are compliant with the overall conservation vision for the respective regions (Biemans *et al*., 2008).

As the existence of the vast majority of species, including endangered species, at least within Europe, depends on areas managed for agriculture and forestry (Tscharntke *et al*., 2005), prohibiting bioenergy production in protected and nature conservation areas alone would not be sufficient as a strategy to prevent negative effects of biomass crop cultivation on biodiversity at the landscape or regional scale. Strategies for sustainable cultivation of biomass crops within production landscapes are therefore needed. An important question in this context, that needs to be addressed by research, is to what degree is biomass production in a certain area related to the (potential) biodiversity value of the same area if reserved for nature or managed by high nature value farming (Dornburg *et al*., 2010)?

One of the essential prerequisites for formulating recommendations are assessments of realistic capacities of landscapes to produce domestic bioenergy feedstock for bioenergy and to avoid over-optimistic projections about the potential contribution of biomass to the energy mix (Florin & Bunting, 2009). Biemans *et al*. (2008) recommended making land inventories which explore the opportunities and necessary restrictions concerning the cultivation of biomass for bioenergy purposes. Equally important for projecting future regional distribution of biomass crops is an understanding of the development of the bioenergy markets and the context of energy production, i.e., smallholder production for local use as against large-scale centralized production for transregional use. Cross-disciplinary research, development of environmental metrics, closing the knowledge gaps regarding positive or negative impacts on biodiversity and integrated modelling approaches are necessary to evaluate sustainability and economic and environmental tradeoffs (Graham, 2007; Firbank, 2008; Groom *et al*., 2008).

The national and international policy dimension. If the full potential of bioenergy is to be realized without causing the associated risks to biodiversity and the environment to occur, there is a pressing need to better understand the full environmental impact throughout the life cycle of bioenergy utilization (Florin & Bunting, 2009; Hennenberg *et al*., 2009). Given the shortcomings of existing sustainability standards in Europe and the USA (see Hennenberg *et al*., 2009 for an overview) development and implementation of mandatory standards and certification systems are needed. However, for the coarse scale impacts in particular, significant uncertainties exist. Risk-based approaches to decision-making, full life-cycle assessments (LCA) or environmental impact assessments (EIA) to assess the net direct and indirect impacts of land-use change could be helpful tools for the development of legally binding regulations (Firbank, 2008; Biemans *et al*., 2008; Florin & Bunting, 2009). As impacts of energy crop production vary on spatial scales finer than the ones usually considered in LCAs, some challenges regarding the implementation of spatially explicit information in order to integrate biodiversity aspects into LCAs have to be overcome (Urban *et al*., 2008). To meet those challenges and to reduce uncertainties, biodiversity research has to contribute knowledge to the risk assessment and decision-making process. In order to make risk and/or sustainability assessments of bioenergy production operable, it is crucial to identify critical criteria, but at the same time keep the number and measurement at a reasonable level (Donnelly *et al*., 2006; Buchholz *et al*., 2009). Otherwise, the aim to integrate biomass crops into agricultural landscapes for stimulating beneficial effects on biodiversity and ecosystem services might be at risk as small and independent producers could be locked out and the market for sustainable biofuels would be then dominated by international investors, most probably resulting in large-scale plantations (Zah *et al*., 2009).

Existing subsidy programmes and incentives for bioenergy and for agri-environment schemes need to be re-evaluated and perhaps integrated for the purpose of sustainable energy crop production (Paine *et al*., 1996). The efficiency of standards, when applied voluntarily or only referring to cross-compliance and good agricultural practice, have fundamental limits in their contribution to sustainable production and in stopping the loss of biodiversity in agricultural production areas (Henle *et al*., 2008; Kaphengst *et al*., 2009). Therefore, governance structures that set clear rules and incentives for biomass crop production and use of natural resources are essential for a more sustainable development (Kaphengst *et al*., 2009). At the same time, risk-mitigation strategies should ideally remain flexible with regard to the various geographical peculiarities, feedstock produced and technologies applied and should use the principles of adaptive management so that policies can be revised as new scientific knowledge emerges (Florin & Bunting, 2009; Hennenberg *et al*., 2009).

Conclusions

The pressing need to mitigate global climate change has resulted in policy makers setting targets for bioenergy production which, for example in the case of bioethanol production from first generation energy crops, has led to some unsustainable production systems and turned out to be ineffective in reducing GHG emissions. Similar trends of unsustainable planning and targeting are apparent for second generation biomass crops in many countries but there is a chance for scientific knowledge to catch up with decision making to ensure strategic and sustainable energy crop production. Many studies on biomass crops in temperate regions report positive effects on biodiversity but they strongly depend on the respective field-scale management of the crops. There is a major concern that large-scale intensive commercial production of biomass crops could have overall negative effects on biodiversity, in particular in areas of high nature-conservation value. Given the vast extent of land that would have to be converted to biomass crops in order to reach the global climate change mitigation targets, the spatial layout and distribution of biomass plantations in landscapes and regions is an essential determinant for the effect of biomass crops on biodiversity and ecosystem services. Biodiversity and ecosystem services would therefore have to become an essential part of risk assessment measures and sustainability appraisals which would have to become an obligatory part of strategic landscape planning to ensure sustainable and environmentally friendly biomass crop production. Taking regional peculiarities into account, risk assessment would have to include coarse scale ecological patterns and processes. To facilitate this, interdisciplinary research and integrated modelling of environmental and economic issues would be necessary to formulate standards that help to support long-term economic and ecological sustainability of bioenergy production and to avoid costly mistakes in our attempts to mitigate global climate change.

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Supporting Information

Additional Supporting Information may be found in the online version of this article:

Appendix S1. List and short description of field-scale studies on effects of biomass crops on diversity, abundance and/or species composition of a range of taxa. Age: if not otherwise stated within the table figures resemble years; Use: C = commercial, D = demonstration, E = experimental, CRP = Conservation Reserve Programme.

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