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REVIEW ARTICLE

Management swing potential for bioenergy crops

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Abstract

Bioenergy crops are often classified (and subsequently regulated) according to species that have been evaluated as environmentally beneficial or detrimental, but in practice, management decisions rather than species per se can determine the overall environmental impact of a bioenergy production system. Here, we review the greenhouse gas balance and 'management swing potential' of seven different bioenergy cropping systems in temperate and tropical regions. Prior land use, harvesting techniques, harvest timing, and fertilization are among the key management considerations that can swing the greenhouse gas balance of bioenergy from positive to negative or the reverse. Although the management swing potential is substantial for many cropping systems, there are some species (e.g., soybean) that have such low bioenergy yield potentials that the environmental impact is unlikely to be reversed by management. High-yielding bioenergy crops (e.g., corn, sugarcane, *Miscanthus*, and fast-growing tree species), however, can be managed for environmental benefits or losses, suggesting that the bioenergy sector would be better informed by incorporating management-based evaluations into classifications of bioenergy feedstocks.

Keywords: biofuel, corn, greenhouse gas, legume trees, mallee, *Miscanthus*, oil palm, soybean biodiesel, sugarcane, *Zea mays*

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Introduction

Bioenergy development is intended, in part, to offset greenhouse gas (GHG) emissions to the atmosphere, but analyses of the GHG balances of bioenergy crops have concluded vastly different magnitudes of this benefit (Davis *et al.*, 2009). Some analyses indicate that the GHG emissions associated with bioenergy agriculture can be significant, thus negating the intended mitigation (e.g., Crutzen *et al.*, 2008; Searchinger *et al.*, 2008), and others find substantial emissions savings compared with fossil fuel use (e.g., Farrell *et al.*, 2006; Adler *et al.*, 2007). These diverse findings are the result of differing assumptions about the geography, system boundaries, inventories, and land use associated with a given bioenergy cropping system (Cherubini *et al.*, 2009; Davis *et al.*, 2009), but real differences among species and management regimes are also clearly evident (Melillo *et al.*, 2009; EPA, 2010; Bird *et al.*, 2011).

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Both the production and environmental impacts of each species considered for bioenergy are dependent on a suite of management decisions. For example, the choice between conventional tillage and minimum tillage in annual cropping systems affects the fossil fuel energy used in farm operations, and can influence the production potential (i.e., yield) and the amount of soil organic carbon (SOC) that is retained (Sisti *et al.*, 2004; Kim & Dale, 2005; Adler *et al.*, 2007; Olchin *et al.*, 2008). For every bioenergy feedstock, there are management considerations that can aid or inhibit the potential of the crop to mitigate GHG emissions. We call this management effect the 'management swing potential', and assert that this is a key component of bioenergy development that deserves more attention when assessing the sustainability of bioenergy.

Here, we first review the traits of biomass species that are generally viewed as favorable for bioenergy. We then explore how these characteristics can be affected by management choices. Finally, we review six case studies, three in temperate regions and three in tropical regions, that exemplify the management swing potential for a diverse range of bioenergy cropping systems, and then explore one case study where a management swing potential was not evident.

Favorable traits of candidate bioenergy crop species

Differences among species in potential benefits for GHG mitigation have become a prominent topic of debate in both scientific literature and policy decisions that address the impact of bioenergy crops (Heaton *et al.*, 2008a; EPA, 2010; Hertel *et al.*, 2010; Davis *et al.*, 2012). The GHG emissions of a bioenergy production chain depend in part on species choice because some species have greater physiological limitations and require more intensive management (with greater inputs) than others. Fertilization requirements, for example, can be an important determinant of GHG emissions in the form of N_2O (e.g., Smeets *et al.*, 2009). Some species also inherently have a greater potential to sequester SOC than others (Sartori *et al.*, 2006; Anderson-Teixeira *et al.*, 2009; Davis *et al.*, 2012).

As the knowledge of bioenergy crop alternatives mounts, it has become evident that there are generalized traits common to many of the species that are viewed as the most viable and renewable options for bioenergy. The primary characteristic that maximizes energy yield is biomass production; very simply, high-yielding crops produce more biomass that can be used for solid or liquid fuel. Plant yield potentials vary by region; thus, 'high yield' is best defined relative to other crops that would grow in similar conditions. Greater biomass yields result in a smaller land footprint per unit of energy product. In addition to high yields, the following traits are also favorable for bioenergy:

- *Low input requirements*

Biomass sources that require low amounts of fertilizer, herbicide, pesticide, farm machinery, and irrigation reduce the energy inputs and GHG costs of agricultural production.

- *Low soil emissions*

Crops that inherently have low rates of decomposition and denitrification that contribute to GHG emissions have a lower environmental impact.

- *Low cost*

Crops that can be grown at a low cost increase the profit margin of the biomass used for bioenergy and reduce the risk associated with investment in bioenergy production.

- *Easy establishment*

The ease with which land can be prepared for a bioenergy crop can substantially affect the environmental impact of that crop. If land preparation, farm equipment and machinery, labor, and establishment time of a crop are all minimal, there is a greater likelihood of optimizing yield and minimizing environmental impact (through energy use, GHG emissions, erosion, nutrient loss, and soil disturbance). Crops that are well understood and already widely produced (e.g., corn in the United States) have been adopted quickly for bioenergy because there was less risk in the establishment phase than there would be for a new crop.

- *Tolerance of extreme and/or variable environments*

Plant tolerance to extreme or variable environmental conditions reduces the risk of yield losses. Plants that can tolerate drought, for example, can potentially be grown in regions where other crops cannot, thus reducing land competition. Other plants can tolerate extreme temperature fluctuations (e.g., frost tolerance) and therefore carry less risk of crop damage that can lead to yield and quality losses.

- *High nutrient-use efficiency (NUE) and/or water-use efficiency (WUE)*

Crops that are highly efficient at recycling nutrients or use less water per unit of production are less likely to require fertilizer or irrigation, and thus reduce cost, energy requirements, and GHG emissions per unit of biomass. A specific consideration is the nutrient content of the harvested biomass (high in the case of fruits or seeds, lower in the case of vegetative parts, lowest where starch-rich storage tissues are harvested).

- *Ecosystem service provisions*

Crop systems that retain C and other nutrients; provide habitat for wildlife; or improve soil, air, and/or water quality by definition reduce environmental impacts by maintaining or increasing ecosystem services.

- *Coproducts*

Bioenergy crops that yield coproducts or by-products from harvest and/or processing biomass can enhance profitability, and also provide environmental benefits by reducing the energy that is required to generate the product through an independent production chain. For example, bagasse that is a coproduct of sugarcane processing for biofuel is often used for heat and power in fuel conversion plants. Without this coproduct, heat and power would have to be generated from another source.

There are few bioenergy crops that have all of the characteristics described above, but a cropping system that is high yielding for a given region and inherently has a combination of any of these traits is more likely to emerge as a suitable bioenergy candidate than one that lacks most of these qualities. Examples of crops that are already in use or heavily researched for bioenergy are provided in Table 1 with relevant characteristics identified.

Many policies that govern bioenergy have been decided based on evaluations of the environmental costs and benefits associated with species that are proposed or currently in use for bioenergy (e.g., RFS2; EPA, 2010). There are fewer policy delineations that are based on the characteristics of cropping systems alone. While species designations are easily transferable categories, the characteristics of management strategies may prove to be more important metrics for classifying bioenergy alternatives.

In contrast to the species-based classification that has been adopted in bioenergy legislation, the forestry sector has produced standards that guide management and harvesting practices (*e.g.*, Forest Stewardship Council, 2010; Manomet Center for Conservation Sciences, 2010). One key consequence of this focus on management was an awareness of the significant potential for deforestation to contribute to C emissions or, inversely, the potential for managed forests to accumulate C. Policies that govern the bioenergy sector might benefit from emulating the forestry sector regulations that focus on management instead of categorizing crops by species.

Our synthesis of the literature provides estimates for the effects of management on GHG emissions, but these estimates are constrained by assumptions that will only be overcome with additional measurements in bioenergy cropping systems as they grow across the global landscape. Effects on SOC will be reviewed in many of the case studies included here because increasing SOC can result in greater sequestration of atmospheric CO_2, but the uncertainty associated with estimated changes in SOC must be acknowledged. This uncertainty is discussed in detail in a case study of palm oil, but similar biases apply to other cropping systems as well.

How can management increase or decrease the environmental impact of a crop?

Management choices can define the characteristics of a cropping system as much as species differences. Each of the traits that were outlined in the previous section either dictates management needs or is determined by a management choice. For example, a crop that has lower NUE may require greater N fertilization to achieve yields similar to that of a crop with high NUE. Nitrogen fertilizer may be a necessity for the survival of the plant, but in some cases fertilization is a management choice that maximizes the crop yield while potentially compromising other environmental benefits (e.g., Renouf *et al.*, 2008).

Management affects the viability of cropping systems for bioenergy through feedbacks to the traits defined above. Management choices occur at different spatial scales; some choices are made at the regional scale, such as whether a forest can be cleared for agriculture, whereas others, such as crop husbandry practices, are made at the field scale. Some examples of management choices that are relevant at both scales are as follow:

- *Previous land use at site of crop establishment*

If the time-averaged C stock of the bioenergy production system, above- and belowground, is greater than

Table 1 Characteristics of bioenergy crops generally associated with net environmental benefits

Crop	High yield	Low input	Low emission	Low cost	Easily established	Tolerant	High NUE	Ecosystem services	Coproduct
Corn grain	✓			✓	✓				✓
Perennial grasses	✓	✓	✓			✓	✓	✓	
Oil palm	✓				✓				✓
Rapeseed									
Residues		✓	✓	✓	✓				✓
Soybean		✓	✓	✓	✓		✓		✓
Sugarcane	✓	✓	✓	✓	✓		✓		✓
Waste	✓	✓	✓	✓	✓				✓
Wheat				✓	✓	✓			✓
Trees	✓	✓	✓			✓	✓	✓	✓

that of the preceding land use, the first production cycle (s) (depending on the accounting rules) can be credited with C sequestration; if it is lower, a 'C debt' needs to be repaid by GHG emission savings from bioenergy use. Negligible or negative C debts avoid pay-back times incurred by land-use change to biofuel crops.

- *Biomass removal*

The amount of biomass removed is directly related to the yield of a cropping system (Dale *et al.*, 2011), but also influences the subsequent input requirements, harvesting energy requirement, emissions from the system, ecosystem services, and cobenefits that can be produced.

- *Drainage*

The way soil is managed to control plant available water can have a very strong effect on yield, emissions, and ecosystem services. Installation of effective drainage can determine the ease with which a particular cropping system can be established.

- *Tillage*

Soil disturbance caused by tillage affects the soil C stocks as well as the fossil fuel energy input required (due to the energy demand of mechanized tillage).

- *Cover or rotating crops*

Cover crops can yield coproducts while also affecting soil C sequestration and emissions associated with soil respiration during fallow periods, wildlife habitat, soil quality, and erosion. Rotating crops can also affect these traits, with the greatest influence often on soil quality.

- *Nutrient management*

The amount and timing of fertilizer or irrigation water that is applied to cropping systems can have a strong influence on emissions (especially N_2O emissions, where complex interactions with stubble retention have been described (Wang *et al.*, 2011)).

- *Harvest frequency and intensity*

The timing and intensity of harvest can affect yields, nutrient use, soil quality, emissions, and cobenefits. Timing is most applicable to nonannual cropping systems that can be harvested at different intervals or in different seasons. Intensity is applicable to the amount of disturbance required at a location and is inversely related to the land area required for production.

It is clear that management decisions can affect the characteristics of a bioenergy cropping system to either enhance the biomass production system so that it becomes favorable for bioenergy or inhibit the environmental benefits of biomass production. In this way, management has the ability to 'swing' the traits of a cropping system toward net benefits (*e.g.,* reduction in GHG emissions), or toward environmental impacts (*e.g.,* increased GHG emissions).

Following, we present case studies of diverse bioenergy cropping systems to demonstrate the magnitude of the management swing potential for different systems. The potential effect of management on GHG fluxes of each case is summarized from previously published literature, and reflects the difference in the net GHG emissions estimated for a bioenergy production chain. The assumptions of life-cycle estimates vary for the different case studies (Table 2), but the effect of a single management change on life-cycle emissions is clearly evident. The magnitude of the management swing potential for each case is summarized in Fig. 1.

Temperate case study 1: Miscanthus

Miscanthus can grow for 15–30 years without having to be replanted. It is highly efficient in recycling nutrients because the plants senesce prior to harvest and nutrients are translocated to underground storage organs. Peak autumn yields of mature stands of *Miscanthus* range from 14 Mg ha^{-1} in the United Kingdom to 50 Mg ha^{-1} in warmer latitudes of the United States (Heaton *et al.*, 2008b). The current practice is to delay harvest until late winter or early spring to obtain higher dry matter content (reduced moisture) and lower mineral content. Delayed harvest results in a reduction in yield, due to litterfall, of around 23–53% depending on location and harvest time (Lewandowski & Kicherer, 1997; Lewandowski *et al.*, 2003). This management choice increases the C that is ultimately sequestered in the soil because the litterfall that occurs between the time of peak aboveground biomass and the time of harvest contributes C to the soil.

The perennial nature of *Miscanthus* is in contrast to annual crops, where the soil is usually plowed at least once per year (except in no-till and minimum tillage systems which still occupy limited areas globally; Smith *et al.*, 2008). Soil disturbance can encourage oxidation and erosion of soil organic matter, so the production of perennial energy crops like *Miscanthus* results in a lower rate of soil C loss than annual crop production. *Miscanthus* also tends to add more litter to the soil than annual crops (as aboveground biomass remains all year round rather than during a specific crop season), and the litter tends to be more recalcitrant (Hastings *et al.*, 2009a,b).

Table 2 Assumptions of life-cycle emissions of GHG reported in Figs 1 and 3

	Miscanthus	Corn	Mallee SRC	Sugarcane (Brazil)	Sugarcane (Australia)	Oil palm
Source	Hillier *et al.*, 2009a;	Kim & Dale, 2005;	Yu & Wu, 2010;	Dias de Oliveira *et al.*, 2005;	Renouf *et al.*, 2008;	Choo *et al.*, 2011;
Life-cycle inventory						
Upstream manufacturing costs	✓	✓	✓	✓	✓	✓
Direct inputs for cultivation	✓	✓	✓	✓	✓	✓
Harvest/baling/transport	✓	✓	✓	✓	✓	✓
Processing		✓	✓	✓		✓
Conversion		✓		✓		✓
Transport		✓		✓		
Coproducts		✓				✓
End use		✓				✓
Fossil fuel displacement	✓			✓		
Land-use change	✓					
Assumptions						
Time horizon for analysis	1–5 years	40 years	50 years	1 year	4–6 years	25 years
Fuel product	Biomass for combustion	Ethanol	Bioslurry for combustion	Ethanol	Ethanol	Biodiesel
Estimated total GHG emissions (CO_2 $ha^{-1}y^{-1}$) with conventional practice	−4.4	1.7	1.2	3.9	0.027	4.5
Management swing potential						
Conventional management*	Planting on land in current or abandoned agriculture	Annual tillage	Harvested every 3 years in spring	Preharvest burning	Preharvest burning	Fronds and empty fruiting bodies removed at harvest, plantation long established (no peatland replacement)
Swing management*	Planting on native forestland	No-till	Harvested every 4 years in fall	Green harvest	Green harvest	Fronds and empty fruiting bodies recycled to soil (plantation long established)
Direction of swing*	+	–	–	–	–	–

*The baseline for miscanthus as shown in the Figs 1 and 3 is not the common conventional practice, and is thus represented differently in this table.

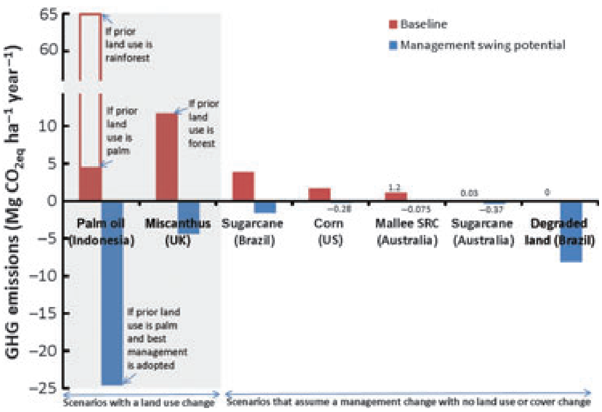

Fig. 1 Management swing potential (blue) for oil palm in Indonesia (if rainforest is not replaced and fronds and empty fruiting bodies are recycled to soil; Fairhurst & Mutert, 1997; Tan *et al.*, 2009; Choo *et al.*, 2011), miscanthus in the United Kingdom (if planted on previously cropped land instead of native forestland; Hillier *et al.*, 2009a), sugarcane in Brazil (Dias de Oliveira *et al.*, 2005) and Australia (if no preharvest burning is practiced, Renouf *et al.*, 2008), corn in the United States (if no-till is practiced; Kim & Dale, 2005), mallee SRC in Australia (if 4 years fall harvest rotations are implemented; Yu & Wu, 2010, Peck *et al.*, 2011), and sloping degraded land in Brazil (if planted with FGLT) relative to a baseline scenario without the swing management (red).

These factors combine to increase SOC stocks for a period of time until the C inputs from *Miscanthus* come into balance with the losses of C from the soil. Carbon dioxide (CO_2) losses under perennial crops are thus reduced relative to annual crops (Dondini *et al.*, 2009; Hillier *et al.*, 2009a; Davis *et al.*, 2012).

Greenhouse gas emissions from 'next generation' perennial lignocellulosic energy crops are on average lower than emissions from annual crops, as almost all annual crops (except legumes) require significant input of fertilizer nitrogen (Williams *et al.*, 2006; Hillier *et al.*, 2009b), leading to greater nitrous oxide (N_2O) emissions; N_2O is 310 times more potent as a GHG than CO_2 (Forster *et al.*, 2007). In contrast, perennial energy crops tend to require less nitrogen fertilizer less often (St Clair *et al.*, 2008). Some studies in the United States have concluded that *Miscanthus* has extraordinarily high NUE (Heaton *et al.*, 2009) and that the nitrogen budget is supplemented by nitrogen-fixing bacteria (Davis *et al.*, 2010). The standard fertilizer regime that is used for similar grass crops could inadvertently overfertilize *Miscanthus*, negatively affecting the GHG balance of the crop.

The GHG emissions of *Miscanthus* also depend on the previous land use at the location where the crop is grown. St Clair *et al.* (2008) estimated annual GHG emissions for annual crops (winter wheat and oil seed rape) and for two lignocellulosic perennial crops (*Miscanthus* and short-rotation willow). The energy crops under typical management resulted in GHG emissions of 0.4–0.5 Mg CO_2-eq. ha^{-1} yr^{-1}, compared with emissions in excess of 2 Mg CO_2-eq. ha^{-1} yr^{-1} for oil seed rape and winter wheat under typical management. Thus, GHG emissions from annual crops were over five times greater than emissions from the perennial energy crops (St Clair *et al.*, 2008). When planted on arable land, *Miscanthus* is a net sink of GHG (−4.4 Mg CO_2-eq. ha^{-1}, Hillier *et al.*, 2009a). Davis *et al.* (2012) demonstrated that land-use change from annual corn cropping for biofuel to perennial grasses for biofuel could transition the Midwestern corn-belt region in the United States from a source to a sink of GHG due to the large increase in soil carbon that would result from planting *Miscanthus*. In contrast, land-use change from native perennial vegetation may incur a C debt and the GHG balance can be unfavorable, emitting up to 12 Mg CO_2-eq. ha^{-1} in the case of changing forestland to *Miscanthus* (Hillier *et al.*, 2009a; Fig. 1). However, relative to perennial bioenergy systems, the replacement of similar land with annual cropping results in even greater GHG emissions (St Clair *et al.*, 2008; Hillier *et al.*, 2009a; Gelfand *et al.*, 2011).

Although *Miscanthus* is associated with net environmental benefits in agricultural landscapes, there is the potential for a negative management swing if management involves the replacement of an existing perennial ecosystem, overfertilization, or early harvesting. Forests and native grasslands can be large C stocks, and although the annual rate of C sequestration may be less in a native ecosystem than a *Miscanthus* crop, the amount of C displaced during land-use change can be large (Fig. 1). Furthermore, overfertilization can lead to N_2O emissions and reduce the inherent NUE of *Miscanthus* by inhibiting the establishment of N-fixing microbial communities, whereas early harvesting can reduce C accumulation in the soil and increase the energy input required to dry the harvested biomass.

Temperate case study 2: corn ethanol

Corn grain is the main source of ethanol in the temperate United States (Wang *et al.*, 2007). Although some studies have shown a reduction in GHG emissions from replacing gasoline with corn-grain ethanol (e.g., Adler *et al.*, 2007), the merit of using corn-based ethanol for emissions reduction remains in question, particularly if there is an initial carbon debt (e.g., Fargione *et al.*, 2008). A carbon debt can occur due to (1) direct land-use change if, for example, forestland is cleared to grow corn for biofuel or (2) indirect land-use change, that is the clearing of land for corn to offset the loss of traditional corn commodity production that was replaced with grain for bioenergy production (Searchinger *et al.*,

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2008). The large amounts of nitrogen fertilizer that are used in corn cropping systems are another large source of GHG emissions. Regardless of the controversy, corn grain is a major feedstock for bioenergy production in the United States, and will likely continue to be a key source of transportation fuel until later generation feedstocks are available for mass production of bioenergy.

Several management options are available for reducing the GHG footprint of corn production systems. Alternative management options include less intensive tillage practices, improved fertilizer management, and residue conservation (Smith *et al.*, 2008). No-till has been estimated to increase carbon sequestration over 20 years by 44 g m^{-2} yr^{-1} with continuous corn, and 90 g m^{-2} yr^{-1} if grown in rotation with soybeans (West & Post, 2002). When assessed in the context of a bioenergy production system, no-till management can 'swing' the impact of producing corn ethanol from a net source of GHG to a net sink of GHG (Kim & Dale, 2005; Fig. 1). Improving agronomic practices can also increase carbon sequestration. Examples of such improvements in corn production systems include (1) adding hay or pasture in the crop rotation, (2) optimizing fertilization rates, (3) limiting bare fallow, and (4) planting more productive corn varieties. Estimates of C sequestration under these management regimes range from 9 to 24 g m^{-2} yr^{-1} over a 20-year time period (Smith *et al.*, 2008). The continual development of higher yielding corn varieties over the last half of a century has led to a fairly constant increase in soil C stocks (Buyanovsky & Wagner, 1998). There is however a maximum SOC that will eventually be reached even with improved management decisions. It should also be noted that if increasing yields are associated with increasing N fertilization, this can enhance N_2O emissions.

Nitrous oxide emissions from soils in corn production systems are high due to high N-fertilizer input, which limits the net GHG benefit of using corn-grain ethanol as a replacement for fossil fuel (Crutzen *et al.*, 2008; Ogle *et al.*, 2008). There are, however, practices that can limit N_2O emissions from corn production. For example, Akiyama *et al.* (2010) reviewed the literature for practices that enhance the efficiency of N-fertilizer use in crops, and found that nitrification inhibitors reduce soil N_2O emissions on average by 38%, and polymer-coated fertilizers reduce emissions by 35% on average. Reducing N_2O emissions can have a significant effect on the GHG footprint of a bioenergy crop that requires significant N-fertilizer inputs.

Temperate case study 3: short-rotation coppice trees

Increasing the planting of perennial vegetation in cleared agricultural landscapes in Australia is being promoted in pursuit of environmental objectives including biodiversity conservation, dryland salinity mediation, and C sequestration (Roberts *et al.*, 2009; PMSEIC, 2010). Large-scale implementation is dependent on these systems being profitable (Bell, 2005), and biomass production is suggested as a possible income source. Many native shrub and tree species have been examined and field tested and are considered suitable for biomass production in Australia including *Eucalyptus* spp., *Acacia* spp., and *Atriplex nummularia* (Olsen *et al.*, 2004; Bennell *et al.*, 2009). In this case study, we focus on the opportunities for coppicing mallee species (e.g., *Eucalyptus polybractea*, *E. loxophleba* ssp *lissophloia*, and *E. kochii*) to be grown as a dedicated energy crop. These crops have the capacity to provide significant biomass with the potential to displace up to 15% of Australia's gasoline (as ethanol) or 9% of current electricity generation (Farine *et al.*, 2012). Mallee species are not suitable for solid wood products, and grow in environments unfit for commercial plantation timber species; so they do not compete with conventional forestry for land resources. Besides biomass for energy production, mallees can also be used for biochar and/or to produce co-products including eucalyptus oil (Hobbs & Bartle, 2009; George & Nicholas, 2012).

Mallees are a hardy group of small multistemmed Eucalyptus tree species found in low rainfall areas across southern Australia (Olsen *et al.*, 2004). Many mallee species have a significant capacity to coppice, enhancing their potential role in biomass production systems where they can be established and then regularly harvested without replanting. Their capacity to survive and grow with limited inputs in dry landscapes led to significant interest in intercropping mallees with cereal crops where salinity was evident (Bartle, 2009).

Mallee trees are commonly planted in belts of two to four rows that are spaced at intervals wide enough to accommodate cereal crop planting and harvesting equipment. As with any crop, growth rates of mallees are determined by the species, soil, rainfall, and nutrient availability. Significant variation in yields within and between sites has been reported in the literature, with the productivity of 4-year-old coppiced stands ranging from 10 to 100 Mg ha^{-1} (Peck *et al.*, 2011). In dryland areas of eastern Australia it is likely that yields will be at the lower end of the spectrum and often less than 10 Mg ha^{-1} yr^{-1} (Milthorpe *et al.*, 1994); this is still high relative to other species grown in the region.

Growing short-rotation coppice (SRC) trees as an intercrop may lead to competition for limited resources. It is clear that crop or pasture yields decline with proximity to mallee belts due to the competition for water and nutrients, although the yield decline may be mitigated by benefits of shelter from wind. Across multiple sites in the Western Australian wheat belt region, Peck *et al.* (2011) and Sudmeyer *et al.* (2012) reported an average competition zone between mallee SRC and the cropping area that extended 10–12 m from the edge of the mallees. The average yield reduction for the cereal crop across this zone was ≈50% in areas adjacent to unharvested mallees. The area of competition reduced to 8 – 9 m when the mallees were regularly harvested. The reduction in competition was dependent on the frequency and seasonal timing of harvest, with competition increasing 3 years after harvest. When harvesting occurred in the fall season, there was a reduction in competition between the mallees and the crops predominantly due to the increased soil water availability during the following winter (Sudmeyer *et al.*, 2012). The pattern and amount of water consumed during biomass production will vary not only between species but also planting regimes (Wildy *et al.*, 2004). Current recommendations are to grow narrow (two row) belts of mallees with wide interrow spacing (4–8 m) to optimize water use and reduce competition between the mallee SRC and cereal crops (Peck *et al.*, 2011). In lower rainfall areas, especially those with variable annual rainfall, Sudmeyer *et al.* (2012) recommend a 4–5 m unplanted buffer between the harvested mallees and other crops to reduce input costs and avoid the zone of strongest competition next to the mallees.

Allowing mallees to grow for 4 years between harvest increases competition with wheat relative to a 3-year harvest system, but it also increases the biomass production and the shelter value. Harvesting mallees at 4-year intervals, in autumn, gives the greatest biomass yield relative to the impact on wheat yield. Adopting this management practice results in a 'swing' of net GHG emissions per unit area (Fig. 1) relative to 3-year harvests in spring. The life-cycle GHG emission of oil mallee intercropped with wheat is based on previously published results (Yu *et al.*, 2009; Peck *et al.*, 2011; assumptions shown in Table 2). We account for the displaced wheat production, i.e., the forgone wheat yield from the area planted to mallees, and from reduced wheat yields in the cropped area adjacent to the tree belts, due to competition from the oil mallees.

The continued harvesting of the coppice mallees will certainly reduce their competition with crops for water. However, in continually removing the biomass it is important that the nutrient status of the soil is maintained and this may require inputs that carry an energy and GHG cost. In forest systems managed for biomass production, the retention of leaf material is recommended (Farine *et al.*, 2012), but all leaf material is removed during harvest of coppicing mallee systems. Thus, there is potentially a need to fertilize (George & Cowie, 2011) and/or include species that have the capacity to fix nitrogen (e.g., *Acacia* spp.) to maintain soil nutrient status (Richards *et al.*, 2010).

The GHG mitigation opportunities from the utilization of woody biomass are significant (Tucker *et al.*, 2009; Froese & Shonnard, 2010), and a major driver for industry development (Bartle & Abadi, 2010). Coppicing mallee systems have the capacity to produce significant amounts of biomass with small GHG emissions (Yu *et al.*, 2009). Yu *et al.* (2008) found that little nonrenewable energy was utilized during the production of coppice mallee biomass (<3% of GHG emissions) and the majority of energy consumed was associated with harvest and transport of the biomass (≈80% of energy inputs). Thus, the low inputs to the mallee SRC system yield a significant return on energy invested. Recent studies indicate an energy return on investment (ratio of output energy: input energy) for coppice mallee SRC systems of >40 (Wu *et al.*, 2008), which compared very favorably with oilseed production in the same environment.

The mallee SRC system provides a good example of the importance and complexity of the management swing potential. The choice of how the plantings are established and maintained and the competition that they create with other crops (e.g., wheat) are all critical aspects of management, especially in relation to competition for water. In designing, implementing, and integrating a mallee SRC biomass production system in low rainfall areas, the management becomes critical to ensure that identified production and GHG goals are achieved. Regular harvesting (e.g., every fourth year, during autumn) of mallee SRC systems, that optimizes the removal of biomass and reduces competition for limited resources (e.g., water), will produce biomass with a greater proportion of leaf material but will also reduce potential competition with other crops. Management will need to be adaptive and innovative to realize a swing potential that achieves a beneficial GHG balance along with other identified goals.

Subtropical–Tropical case study 1: sugarcane

- *Burning vs. green harvest and residue retention*

Burning of a sugarcane crop prior to harvest removes very large volumes of senescent leaves (trash), most of which are attached to the stems and greatly hinder manual cutting. The practice of preharvest burning was introduced in Brazil in the 1940s because of labor shortages and soon became almost universal practice. The smoke and soot produced is not only an agent for global warming but also a public health risk (Godoi *et al.*, 2004; Arbex *et al.*, 2007), and since the 1990s there has been a gradual reduction in sugarcane burning, required by law since 2003 in the State of São Paulo. In

2010 Brazil harvested 9.1 million ha of sugarcane of which approximately 55% was subjected to preharvest burning and manual harvest whereas the remainder was mechanically harvested 'green' (without burning).

National mean cane yield is ~80 Mg millable cane ha^{-1}. With this productivity, 12–15 Mg (dry weight) of trash (unused biomass) is produced (Souza *et al.*, 2005; Luca *et al.*, 2008; de Figueiredo & La Scala, 2011). As a result of burning this trash (~45% C), emissions would be less than 20–25 Mg ha^{-1} of CO_2 (maximum based on 100% conversion efficiency). When all trash is conserved, it will in warm regions of Brazil gradually decompose from one harvest to the next (12 months later), resulting in similar CO_2 emissions. In studies performed on unburned cane in the north east of Brazil, it was estimated that over 95% of the trash present at harvest decomposed before the next harvest 12 months later (Resende *et al.*, 2006). However, in the cooler drier winters of São Paulo, decomposition is much slower. São Paulo State is responsible for almost 60% of cane production in Brazil (mean yield in 2010 of 85 Mg ha^{-1}) on 55% of the area (IBGE, 2012). Campos (2003) working in São Paulo State found that over a 6-year period the quantity of residual trash on the soil surface stabilized at between 4.5 and 5.5 Mg dry matter (2–2.5 Mg C) ha^{-1} in unburned cane. Thereafter, a mean of 13 Mg dry matter ha^{-1} of trash from cane is produced. GHG mitigation thus increases as a result of the change from preharvest burning to trash conservation because of the C in trash that is added to the soil surface.

When preharvest burning is abandoned, the management practice changes from manual harvesting of burned cane to mechanical harvest of green cane. The impact of the use of mechanical harvesters on soil compaction can be severe as was apparent in the higher soil bulk density of unburned plots observed by Souza *et al.* (2005) and Luca *et al.* (2008). Plantation managers know that compaction caused by heavy harvesting machines will reduce yields and inhibit root growth, leading to a reduction in the residual C inputs to the SOC pool.

Based on available data, we estimate that the management change from preharvest burning to green-cane harvesting for most plantations which are on Ferralsols (>60% of all cane plantations) would result in SOC accumulation of 0.5–1 Mg C ha^{-1} yr^{-1} for the first 10 years, and half that rate (0.25–0.5 Mg C ha^{-1} yr^{-1}) for the next 10 years. Subsequent increases are likely to be very slow. Site-specific data will be needed to provide precise estimates of the SOC accumulation rate, which depends on previous history (how many years of burned cane or crop or pasture cultivation occurred before the management change), yield (determined largely by rainfall and pest and fertilizer management), and other edapho-climatic factors such as soil texture.

It is likely that there will be an impact of the switching from burned to green-cane harvesting on N_2O emissions also. In Brazil, N_2O emissions of sugarcane that has undergone preharvest burning are equivalent to ~465 kg CO_2 ha^{-1} (Dias de Oliveira *et al.*, 2005). Australian studies suggest that the removal of trash from the soil surface decreased N_2O emissions by 24–30%, depending on the area studied (Wang *et al.*, 2008). Applying urea fertilizer to the surface trash that is left after a green harvest, instead of applying to the bare soil surface, increases N-fertilizer loss and reduces N_2O emission through denitrification, but in many cases urea inputs are being increased to compensate for this (Lara-Cabezas *et al.*, 1999). The presence of the trash conserves soil water which is likely to increase N_2O emissions and soil compaction (Ball *et al.*, 1999) due to the traction of heavy harvesting machines. On the other hand, the presence of trash has been shown to promote CH_4 uptake by the soil (Weier, 1998).

The management shift from burning to trash conservation has an effect on SOC stocks, soil N_2O, and CH_4 emissions, but there is also a substantial reduction in direct emissions from burning. This reduction is partially offset by an increase in diesel fuel use (mechanical harvesters), but even after accounting for the diesel fuel used by harvesting equipment when cane is harvested green (~40 L to harvest 70 Mg cane h^{-1}), the emissions from burning results in far greater GHG than without burning (Soares *et al.*, 2009). Sanhueza (2009) suggested that the black C (soot) produced during burning could make large contributions to global warming, more than doubling total GHG emissions in the agricultural phase of production. Galdos *et al.* (2010) estimated that this pollutant could be equivalent to an emission of over 5.5 Mg CO_2 eq ha^{-1} which would reduce the bioethanol displacement of fossil fuel emissions from 13.2 (Soares *et al.*, 2009) to 7.7 Mg CO_2 eq ha^{-1}. In view of this, the debate about the contribution of SOC increases to total GHG mitigation when burning is replaced by green-cane harvesting becomes considerably less important.

Changing from preharvest burning to trash conservation has a very significant and favorable impact on GHG mitigation of sugar cane for bioethanol in Brazil. The elimination of gaseous and particulate emissions associated with preharvest burning results in a substantial management swing from net GHG emissions to a net GHG sink with the use of ethanol from sugarcane that replaces fossil fuel (Dias de Oliveira *et al.*, 2005; Fig. 1).

Tropical case study 2: Fast-growing legume trees on degraded lands

Degraded lands have been proposed as ideal for bioenergy feedstock production. In 1990, an FAO report suggested that 27% of all land area in Latin America, and 236 million ha in Brazil, was either severely or very severely degraded (Bot *et al*., 2000). The report defined severely degraded land as that with 'biotic functions largely destroyed; non-reclaimable at the farm level' and very severely degraded as 'biotic functions fully destroyed, non-reclaimable at the farm level'. It is clear that much of the degraded area so classified in Brazil includes areas of planted pastures, which total at least 50 Mha of mostly gently rolling country in the Cerrado region that can be recovered with carefully managed sugarcane and/or production of other crops. There is much degraded land however that will not support conventional crops in the deforested hillsides of the Atlantic coastal region. The steep slopes and relatively high rainfall (>1000 mm), much of it from sudden tropical storms, cause widespread gully and sheet erosion. An alternative management strategy, planting fast-growing legume trees (FGLT), has been proposed to restore degraded hillsides of the Atlantic forest region and produce biomass for bioenergy.

Planting of FGLTs (e.g., species of *Acacia*, *Mimosa*, and *Gliricidia* among many others) in degraded tropical areas that have sloping topography can lead to the recovery of hillsides. If rhizobium-inoculated seeds are utilized, the seedlings are already nodulated, and fixing nitrogen when transplanted. This technology has been widely used in the Atlantic forest region for recovery of degraded land and in Amazonia for revegetating mine wastes and tailings (Chaer *et al*., 2011). Seedlings are also inoculated with arbuscular mycorrhizal fungi (AMF), the spores of which can be almost completely absent in areas where subsoil is exposed by machinery or severe erosion. In areas within the Atlantic forest region of Brazil where rainfall is abundant and there is no marked dry season, full ground cover is usually attained 18 months after planting. Experience has shown that the key to the success of this technology is the prior inoculation of the tree seeds with selected rhizobium and AMF.

Recent studies have assessed the impact of transplanting FGLT on SOC accumulation of sloped and degraded land. At a site on the coast of the State of Rio de Janeiro near Angra dos Reis, an area where all topsoil had been removed on a steep slope, seedlings inoculated with rhizobium and AMF of the species *Acacia mangium*, *A. auriculiformis*, *Enterolobium contortisiliquum*, *Gliricidia sepium*, *Leucaena leucocephala*, *Mimosa caesalpiniifolia*, and *Paraserianthes falcataria* were planted (Macedo *et al*., 2008). After 13 years it was estimated that SOC stocks to a depth of 60 cm increased by 23 Mg C ha^{-1}, a mean of 1.7 Mg C ha^{-1} yr^{-1}. In another study, degraded pasture land was reclaimed using mixtures of N_2-fixing legume trees grown from preinoculated (rhizobium and AMF) seedlings and nonfixing legumes and nonlegumes (Jensen *et al*., 2011). The mixtures ranged from 0 to 75% N_2-fixing legume trees. In a period of 6 years, the accumulated C in plots with 75% N_2-fixing trees reached 47, 18 Mg ha^{-1} in trunk wood alone compared with only 16 Mg ha^{-1} (4 Mg ha^{-1} in trunk wood) in plots with only non-N_2-fixing species (Fig. 2A). The plots with 50 or 75% N_2-fixing species accumulated 10 Mg ha^{-1} of SOC (0–60 cm) more than those with 0% (Fig. 2B; Jensen *et al*., 2011).

Whereas SOC accumulation is expected to asymptotically approach a limit within 20–30 years, regular harvesting of trunk wood for bioenergy production could mitigate up to 3 Mg C (11 Mg CO_2) ha^{-1} yr^{-1}. At the same time, this FGLT technology provides an array of valuable ecosystem services including reduction in erosion, increased rainfall infiltration, and improved water quality which lead to important reductions in silt

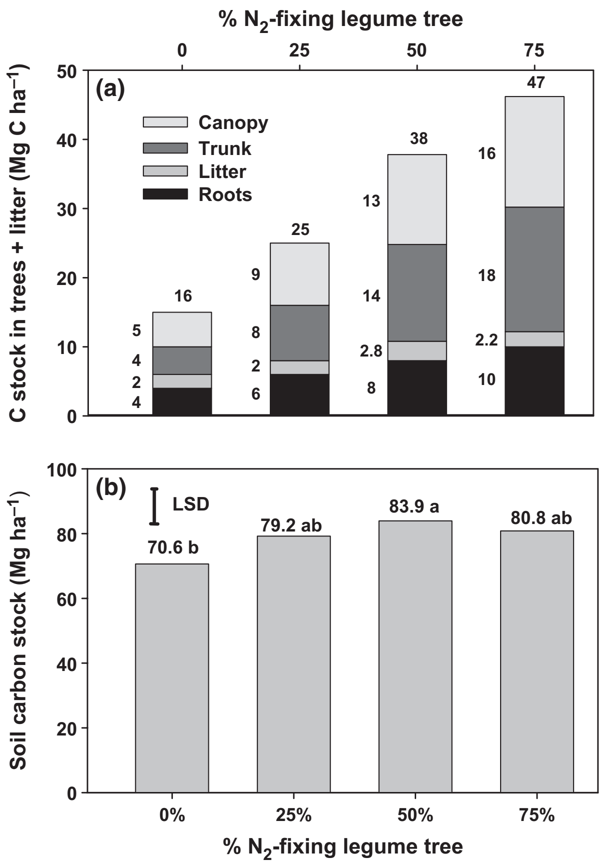

Fig. 2 Total C stocks (A) in biomass and (B) in soil (0–60 cm) under different mixtures of N_2-fixing legume, non-N_2-fixing legume, and nonlegume trees after 6 years of growth on a degraded pasture site (Valença, Rio de Janeiro). Data from Boddey *et al.* (2008). The same letter above the bars indicates that the difference between the means was not significantly different at $P < 0.05$.

deposits to rivers and in flash floods which claim many lives and cost millions of dollars in destroyed farm land and housing every year in many countries of the developing world.

The GHG flux of degraded lands in tropical regions depends on management decisions and there is a management swing potential associated with well-managed FGLT that can be grown for bioenergy in the degraded lands of Brazil (Fig. 1). No life-cycle assessment of the production of this system has yet been completed; the GHG change in this case is based solely on terrestrial fluxes.

Tropical case study 3: oil palm

Oil palm is productive, profitable, and expanding in area (Corley & Tinker, 2003), but its success as an agricultural commodity is reason for environmental concern. Current production of South-East Asian palm oil and use as biofuel is far from C neutral (Reijnders & Huijbregts, 2008; Sheil *et al*., 2009; Reijnders, 2011). Use of peat soils causes CO_2 emissions that exceed the amount sequestered in harvested products (Couwenberg *et al*., 2010; Hooijer *et al*., 2010; Hergoualc'h & Verchot, 2011); 5% of peat soils in the portfolio of a palm oil production facility is sufficient to reduce the average 'emissions saving' ratio to values below the standards set by the EU Renewable Energy Directive (Khasanah *et al*., 2012). Due to consumer pressure and environmental concerns of major stakeholders in the palm oil value chain, new deforestation and use of peat soils for oil palm has been reduced under voluntary agreements of the Roundtable on Sustainable Palm Oil (http://www.rspo.org/; Tan *et al*., 2009; Laurance *et al*., 2010; Choo *et al*., 2011). Land-use change from low-C-stock vegetation on mineral soils is seen as the future of sustainable palm oil, and according to some authors can lead to net gains of SOC stocks (Germer & Sauerborn, 2008; Verhoeven & Setter, 2010; Flynn *et al*., 2011; Hassan *et al*., 2011; Patthanaissaranukool & Polprasert, 2011; Siangiae-o *et al*., 2011).

Environmental sustainability of palm oil is mostly determined by land use or land-use change, and maintenance of soil quality (Parish *et al*., 2008). Land-use change driven by increases in oil palm production areas is the most important factor to consider when analyzing GHG emissions of palm oil production (Parish *et al*., 2008; Stichnothe & Schuchardt, 2011). If peatland is deforested and drained, C and N_2O emissions increase dramatically (Hooijer *et al*., 2006). IPCC assumes N_2O emissions of 8 kg N_2O ha^{-1} yr^{-1} (uncertainty range 0–24 kg) for tropical organic forest soils (IPCC, 2006). Yearly N_2O emissions from a secondary peatland forest in Kalimantan, Indonesia, can be as high as 143 kg ha^{-1} (Couwenberg *et al*., 2010). With the replacement of rainforest and grassland, CO_{2eq} values are estimated to be between 425 and 1850 kg CO_{2eq} Mg^{-1} of fresh fruit bunch production (Fig. 1; Stichnothe & Schuchardt, 2011). It is estimated that it would take between 75 and 93 years for the carbon emissions saved through use of biofuel to compensate for the carbon lost through deforestation. If peatland was replaced, the carbon balancing would take more than 600 years, but can also be infinitely long as annual peat emissions exceed CO_2 savings from biofuel use derived from palm oil on peat soils (Danielson *et al*., 2009).

Estimates of SOC gains in oil palm plantations should be considered with caution as few studies on the biophysical performance of oil palm production after land-use change have been carried out and the literature is based on isolated case studies and unconstrained modeling exercises at best (Adachi *et al*., 2011; Nair *et al*., 2011). Empirical data of both initial SOC and trends over time during a production cycle are needed to verify the claims that C stocks will increase and validate or improve the models used. As part of the Second Assessment Report of the IPCC, Paustian *et al*. (1997) summarized known effects of land-use change on SOC across climatic zones and soil types. Subsequent literature has led to some refinement: Don *et al*. (2011) in a global meta-analysis of 385 studies on land-use change in the tropics found that the highest SOC losses were caused by the replacement of primary forest with cropland (−25%) and perennial crops (−30%), but replacement of forest with grassland also reduced SOC stocks by 12%. If it were a simple additive system, one might thus expect the replacement of grasslands with perennial crops to lead to a decrease in SOC by about 18%. Another recent meta-analysis (Powers *et al*., 2011) focused on 'paired plot' literature and found little consistency in SOC change, with both 'forest to grassland' and 'grassland to forest' transitions leading to statistically significant SOC gain; this suggests a selection bias in the results that get published. Both reviews confirm that complete data sets that combine soil bulk density and SOC are scarce, and that spatial extrapolation is hindered by inadequate representation of tropical soils.

In Sumatra, van Noordwijk *et al*. (1997) estimated that SOC losses occurred when native ecosystems were replaced with croplands. Losses were similar but less than those documented by Don *et al*. (2011) because permanently cropped upland soils are relatively scarce in Sumatra where intensifying land management practices have begun to favor permanent tree crops (van Noordwijk *et al*., 2008). Imperata grasslands and areas formerly used for shifting cultivation may not have substantially lower SOC content than forests (Santoso *et al*., 1997). It thus appears that the expectation that shifting palm oil production to 'degraded forests' or 'grasslands' will lead to a net gain in SOC stocks is unsubstantiated, and the main 'swing potential', apart from avoidance of peat soils, would lie with the aboveground C stock of the preceding vegetation on the land that is chosen for oil palm. With the aboveground time-averaged C stock of oil palm plantations at around 40 Mg C ha^{-1} (Dewi *et al.*, 2009), the threshold for avoiding C debt issues is reached at a 40 Mg C ha^{-1} level on vegetation C maps.

Once an oil palm plantation has been established, good management practices can make a large difference in the greenhouse gas emissions of palm oil production. Compared with other tropical cropping systems, the amount of organic residues, in the form of pruned fronds, returned to soil in oil palm plantations is large, and is estimated to be in the range 10–15 Mg ha^{-1} yr^{-1} of dry matter (DM; Santoso, 1996; Fairhurst & Mutert, 1997). This is more than the 8 Mg ha^{-1} yr^{-1} of DM roughly needed to sustain soil organic matter in humid tropical environments (Fairhurst & Mutert, 1997). A comparison of different management practices in Malaysian oil palm plantations showed that the emissions from production and processing of one ton of fresh fruit bunches can be more than 460 kg CO_{2eq} in the worst case and 110 kg CO_{2eq} Mg^{-1} of bunches in the best case (Stichnothe & Schuchardt, 2011). Cocomposting of empty fruit bunches and palm oil mill effluents and application of the resulting compost back into the field minimizes emissions in the best case.

It is clear that even good management practices cannot compensate for the greenhouse gas emissions resulting from the removal of native forest (Fig. 1). However, once an oil palm plantation is present (which is of course the case now in large areas in South-East Asia), oil palm production scores high on many environmental indicators (Fig. 1; De Vries *et al.*, 2010). Oil palm requires 7–11 times less land area than soybean, rapeseed, and sunflower to produce the same amount of oil.

Other potential 'swing' management practices for palm oil as biofuel feedstock are the use of methane trapping in processing mills and the use of fertilizer to increase yields. Methane traps are technically feasible for new mills, but require investment and integration in the design of processing plants. Eliminating methane emissions would result in a large GHG reduction. Fertilization decisions can also affect GHG emissions of palm oil cropping systems. Depending on the N_2O/N-fertilizer emission ratio that is used in the accounting schemes (Hoefnagels *et al.*, 2010), the additional yield associated with N-fertilizer use may increase or decrease the 'emission footprint' per unit bioenergy derived.

Temperate/tropical crop with no swing potential: soybean for biodiesel

Soybean (Glycine max) is widely consumed as human food and is easy to grow in both tropical and temperate climates. In recent years, especially in Brazil, there has been a push for soybean biodiesel production (Volpi, 2010). Biodiesel and the increasing demand for soybean-based animal fodder are the main causes for the rapid expansion of soybean in Brazil. With an average agricultural yield of 1.5–3.0 Mg ha^{-1} yr^{-1}, an oil content of about 17–20% and an oil production per ha of about 500–700 L, soybean is a modest source of oil (Nogueira, 2010) and has limited environmental benefits relative to other biofuel crops (De Vries *et al.*, 2010).

Soybean is widely used because it is easy to grow. Soybeans have a high NUE due to a symbiotic relationship with rhizobium bacteria that fix nitrogen from the atmosphere: it therefore needs little N fertilizer. Production and transport of N fertilizer can cost a considerable amount of the total fossil energy required for agricultural production. For example, N fertilizer for rapeseed accounts for 65% of the crop's energy requirement. For soybean, energy costs for fertilizer can be as low as 5% of the total energy requirement (De Vries *et al.*, 2010).

Despite the benefits of nitrogen fixation, soybean biodiesel production is strongly dependent on the use of nonrenewable resources in the agricultural production, transport, and industrial processing stages. This leads to a relatively low fraction of the fuel that is actually renewable (around 30%; Cavalett & Ortega, 2009). Research performed in the Londrina region, Paraná State in Brazil concluded that to produce 1 kg of soybean (with 18% vegetable oil) about 1.38 MJ of fossil fuel is required. Assuming a yield of 4 Mg ha^{-1}, this equates to a fossil energy cost for soybean oil of 7.66 MJ kg^{-1} (Gazzoni *et al.*, 2006). A study across four different sites in Brazil estimated an average oil yield of 880 L per ha and an average energy-output/fossil fuel-input energy ratio of 3.3 (Nogueira, 2010). Another Brazilian study assumed a productivity of 552 kg of soybean biodiesel per hectare and estimated 2.30 for the energy-output/fossil fuel-input ratio (Cavalett & Ortega, 2009). In temperate regions of the United States, an output/input energy ratio in the range 3.2–3.4 was estimated for whole grain soybean, but in the output/input energy ratio is only 1.0–1.2 when estimated for the oil product. Soybean has low energy efficiency relative to crops like oil palm. Financially viable biodiesel production can only be achieved if yields are above 2 Mg ha^{-1} (Vera-Diaz *et al.*, 2008). Model studies indicate that only roughly 20% of the Amazon Region (outside of protected areas) can provide yields greater than 2 Mg ha^{-1} (Vera-Diaz *et al.*, 2008).

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These calculations do not include any effect of direct or indirect land-use changes caused by expansion of soybean areas. Expansion of soybean farms is likely to lead to negative direct and indirect impacts of land-use change. However, only a part of these impacts can be allocated to biodiesel production, as animal fodder production is the main driver for soybean growth (Volpi, 2010). It should be noted that biodiesel is usually a coproduct of a system also supplying soy meal.

Soybean exemplifies a crop that has several of the ideal characteristics for bioenergy crops (Table 1), but due to the low yield there is no evidence for a management swing potential that can reverse the emissions of this production system. Management of soybean thus far indicates that there is very little potential to 'swing' the environmental consequences of soybean biodiesel in a beneficial direction.

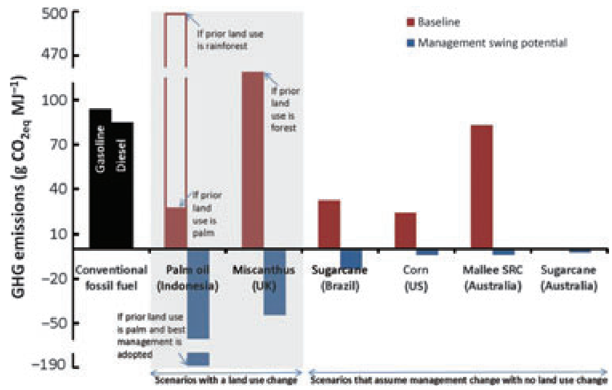

Fig. 3 Management swing potential (blue) for oil palm in Indonesia (if rainforest is not replaced and fronds and empty fruiting bodies are recycled to soil; Fairhurst & Mutert, 1997; Choo *et al.*, 2011), miscanthus in the United Kingdom (if planted on previously cropped land instead of native forestland; Hillier *et al.*, 2009a), sugarcane in Brazil (Dias de Oliveira *et al.*, 2005) and Australia (if no preharvest burning is practiced, Renouf *et al.*, 2008), corn in the United States (if no-till is practiced; Kim & Dale, 2005), and mallee SRC in Australia (if 4 years fall harvest rotations are implemented; Yu & Wu, 2010, Peck *et al.*, 2011) relative to a baseline scenario without the swing management (red) and conventional fossil fuels (black).

Discussion and Conclusion

To address the concomitant problems of climate change and increasing energy demands, a shift from human reliance on fossil fuels to more diversified and renewable energy sources is necessary. Bioenergy is already used in many regions of the world for heat and electricity although conversion of biomass to liquid fuels is also on the rise (Somerville *et al.*, 2010). Species that are currently cultivated for bioenergy vary geographically from palm in the tropics (Wicke *et al.*, 2008) to grasses in temperate regions (DOE, 2006) and coppicing woody tree species in northern and southern latitudes (Weih, 2004). The environmental benefits and risks of feedstock sources vary widely, and can be altered significantly by management choices.

Crop species themselves are neither exclusively positive nor negative, rather it is how they are managed for bioenergy that will determine their impacts. Here, we reviewed examples of the management swing potential for different bioenergy cropping systems. Even though the assumptions used to calculate life-cycle emissions of biofuel production systems vary (Table 2), it is clear that a single management change often has the potential to reverse the net emissions of the entire production chain (Figs 1 and 3). Greenhouse gas emissions can be calculated on an area basis or on an energy basis; GHG emissions per unit energy are more indicative of the mitigation potential and allow for a clearer comparison against the emissions associated with the production and use of conventional fossil fuels (Fig. 3). Only managements that require the replacement of native forest lands result in greater GHG emission per MJ than gasoline or diesel. The production of most bioenergy feedstocks results in lower GHG emissions than fossil fuel production, but certain management choices can actually lead to net negative CO_{2eq} emissions.

The reference case against which bioenergy production scenarios are compared can alter the apparent benefit for GHG reduction. The baseline case can be defined in terms of the previous land management, the previous energy production scenario, or the native ecosystem that would otherwise dominate a landscape. Here, the change in GHG emissions associated with the management swing potential is calculated based on a comparison between a typical bioenergy cropping scenario and the same scenario with a single difference in management. The conditions for the baseline vary among the case studies (Table 2) and it should be noted that a different baseline condition, the initial land use for example, will alter the magnitude of the swing effect estimated here. The oil palm case study provides an example of this where adoption of the best management practices result in a 'swing' of GHG emissions only if the prior land use is not native forest (Figs 1 and 3).

There was evidence of a management swing potential in six of the seven case studies reviewed here. The net GHG sink of *Miscanthus* can be negated by a poor land management decision (Fig. 3), but would also be affected by early harvesting or overfertilization. The net GHG emissions of corn production for biofuel can be offset by management decisions that reduce tillage (Fig. 3), retain residual biomass onsite, or alter fertilization timing and rates. In the case of mallee SRC, the timing of harvest can determine whether the production system is a net source or sink of C (Figs 1 and 3). In tropical regions, the management switch to green-harvested sugarcane without preharvest burning can reverse the climate change impact associated with this cropping system (Figs 1 and 3). Fast-growing legume trees can be used to reclaim sloping degraded lands while capturing and storing C (Fig. 2), but production as a dedicated energy crop has yet to be demonstrated and no life-cycle assessments of this cropping system have been completed. The estimates for FGLT shown in Fig. 1 reflect only the terrestrial C emissions. In the case of palm oil, prior land management and the recycling of fruiting bodies and fronds are critical management considerations that determine whether this energy production system is a net source or sink of GHG.

Not all cropping systems, however, have management options that can 'swing' the environmental impact. Soybean serves as an example of this because, although it has high NUE – a trait favorable for bioenergy crops, it has a low yield relative to other crops and thus the production of soybean incurs a large C footprint per unit area of land.

The international debate about the benefits of biofuels is not likely to be resolved with a generalized view of bioenergy impact assessment because management approaches vary regionally. The case studies reviewed here clearly indicate that a diversified assessment approach is needed to account for many management practices that can swing the overall impact of bioenergy crop production from negative to positive or vice versa. The management swing potential is a key part of the sustainability puzzle, but is underrepresented in the policy debates that will decide the future role of bioenergy in mitigating climate change.

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