IPCC 2014 AR5 WG3 11.13 Appendix Bioenergy: Climate effects, mitigation options, potential and sustainability implications

Important summery quote. BECCS features prominently in many mitigation scenarios. BECCS is deployed in greater quantities and earlier in time the more stringent the climate policy (Section 6.3.5). Whether BECCS is essential for mitigation, or even sufficient, is unclear. In addition, the likelihood of BECCS deployment is difficult to evaluate and depends on safety confirmations, affordability and public acceptance.

11.13.1 Introduction

SRREN (IPCC, 2011) provided a comprehensive overview on bioenergy (Chum et al., 2011). However, a specific bioenergy Appendix in the context of the WGIII AR5 contribution is necessary because (1) many of the more stringent mitigation scenarios (resulting in 450 ppm, but also 550 ppm CO2eq concentration by 2100, see Section 11.9.1) heavily rely on a large-scale deployment of bioenergy with carbon dioxide capture and storage (BECCS); (2) there has been a large body of literature published since SRREN, which complements and updates the analysis presented in this last report; (3) bioenergy is important for many chapters (Chapters 6; 7; 8; 10; 11), which makes it more useful to treat it in a single section instead of in many scattered chapter sections throughout the report. Chapter 11 is the appropriate location for the Appendix, as bioenergy analysis relies crucially on land-use assessments.

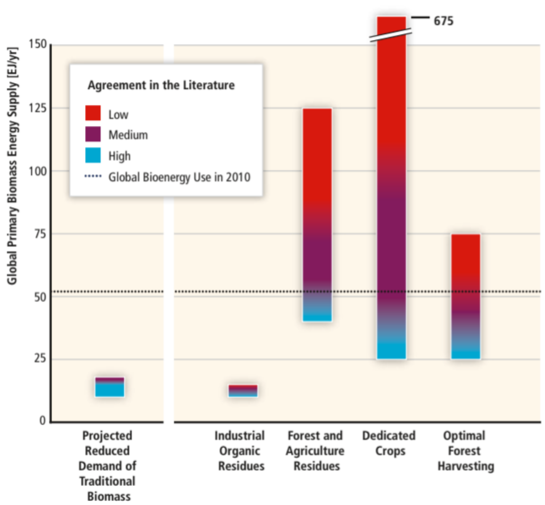
Bioenergy is energy derived from biomass, which can be deployed as solid, liquid, and gaseous fuels for a wide range of uses, including transport, heating, electricity production, and cooking (Chum et al., 2011). Bioenergy has a significant mitigation potential, but there are issues to consider, such as the sustainability of practices and the efficiency of bioenergy systems (Chum et al., 2011). Bioenergy systems can cause both positive and negative effects and their deployment needs to balance a range of environmental, social, and economic objectives that are not always fully compatible. The consequences of bioenergy implementation depend on (1) the technology used; (2) the location, scales, and pace of implementation; (3) the land category used (forest, grassland, marginal lands, and crop lands); and (4) the business models and practices adopted—including how these integrate with or displace the existing land use.

As an update to the SRREN, this report presents (1) a more fine-grained assessment of the technical bioenergy potential reflecting diverse perspectives in the literature; (2) recent potential estimates on technological solutions such as BECCS; (3) an in-depth description of different lifecycle emission accounting methods and their results; (4) a small increase in uncertainty on the future economic bioenergy potential; (5) a comprehensive assessment of diverse livelihood and sustainability effects of bioenergy deployment, identifying the need for systematic aggregation.

11.13.2 Technical bioenergy potential

The technical bioenergy potential, also known as the technical primary biomass potential for bioenergy, is the amount of the theoretical bioenergy output obtainable by full implementation of demonstrated technologies or practices (IPCC, 2011). Unfortunately there is no standard methodology to estimate the technical bioenergy potential, which leads to diverging estimates. Most of the recent studies estimating technical bioenergy potentials assume a 'food/fibre first principle' and exclude deforestation, eventually resulting in an estimate of the 'environmentally sustainable bioenergy potential' when a comprehensive range of environmental constraints is considered (Batidzirai et al., 2012). Recently published estimates that are based in this extended definition of global technical bioenergy potentials in 2050 span a range of almost three orders of magnitude, from 1,000 EJ/yr (Smeets et al., 2007; Field et al., 2008; Haberl et al., 2010; Batidzirai et al., 2012). For example, Chum et al. reported global technical bioenergy potentials of 50–500 EJ/yr for the year 2050 (IPCC, 2011), and the Global Energy Assessment gave a range of 160–270 EJ/yr (Johansson et al., 2012). The discussion following the publication of these global reports has not resulted in a consensus on the magnitude of the future global technical bioenergy potential, but has helped to better understand some of its many structural determinants (Wirsenius et al., 2011; Berndes, 2012; Erb et al., 2012a). How much biomass for energy is technically available in the future depends on the evolution of a multitude of social, political, and economic factors, e.g., land tenure and regulation, trade, and technology (Dornburg et al., 2010). Figure 11.20 shows estimates of the global technical bioenergy potential in 2050 by resource categories. Ranges were obtained from assessing a large number of studies based on a food/fibre first principle and various restrictions regarding resource limitations and environmental concerns but no explicit cost considerations (Hoogwijk et al., 2005; Smeets et al., 2007; Smeets and Faaij, 2007; van Vuuren et al., 2009; Hakala et al., 2009; Dornburg et al., 2010; Haberl et al., 2010, 2011a; Gregg and Smith, 2010; Chum et al., 2011; GEA, 2012; Rogner et al., 2012). Most studies agree that the technical bioenergy potential in 2050 is at least approximately 100 EJ/yr with some modelling assumptions leading to estimates exceeding 500 EJ/yr (Smeets et al., 2007). As stated, different views about sustainability and socio-ecological constraints lead to very different estimates, with some studies reporting much lower figures.

As shown in Figure 11.20, the total technical bioenergy potential is composed of several resource categories that differ in terms of their absolute potential, the span of the ranges—which also reflect the relative agreement/disagreement in the literature—and the implications of utilizing them. Regional differences—which are not addressed here—are also important as the relative size of each biomass resource within the total potential and its absolute magnitude vary widely across countries and world regions. Forest and Agriculture residues. Forest residues (Smeets and Faaij, 2007; Smeets et al., 2007; Dornburg et al., 2010; Haberl et al., 2010; Gregg and Smith, 2010; Rogner et al., 2012) include residues from silvicultural thinning and logging; wood processing residues such as sawdust, bark, and black liquor; and dead wood from natural disturbances, such as storms and insect outbreaks (irregular source). The use of these resources is in general beneficial and any adverse side-effects can be mitigated by controlling residue removal rates considering biodiversity, climate, topography, and soil factors. There is a near-term tradeoff, particularly within temperate and boreal regions, in that organic matter retains organic C for longer if residues are left to decompose slowly instead of being used for energy. Agricultural residues (Smeets et al., 2007; Hakala et al., 2009; Haberl et al., 2010, 2011a; Gregg and Smith, 2010; Chum et al., 2011; Rogner et al., 2012) include manure, harvest residues (e.g., straw), and processing residues (e.g., rice husks from rice milling) and are also in general beneficial. However, mitigating



potential adverse side-effects—such as the loss of soil C—associated to harvesting agriculture residues is more complex as they depend on the different crops, climate, and soil conditions (Kochsiek and Knops, 2012; Repo et al., 2012). Alternative uses of residues (bedding, use as fertilizer) need to be considered. Residues have varying collection and processing costs (in both agriculture and forestry) depending on residue quality and dispersal, with secondary residues often having the benefits of not being dispersed and having relatively constant quality. Densification and storage technologies would enable cost-effective collections over larger areas. Optimization of crop rotation for food and bioenergy output and the use of residues in biogas plants may result in higher bioenergy yields from residues without food-energy competition. Optimal forest harvesting is defined as the fraction of sustainable harvest levels (often set equal to net annual increment) in forests available for wood extraction, which is additional to the projected biomass demand for producing other forest products. This includes both biomass suitable for other uses (e.g., pulp and paper production) and biomass that is not used commercially (Smeets and Faaij, 2007; Chum et al., 2011). The resource potential depends on both environmental and socio-economic factors. For example, the change in forest management and harvesting regimes due to bioenergy demand depends on forest ownership and the structure of the associated forest industry. Also, the forest productivity—and C stock—response to changes in forest management and harvesting depends on the character of the forest ecosystem, as shaped by historic forest management and events such as fires, storms, and insect outbreaks, but also on the management scheme (e.g., including replanting after harvest, soil protection, recycling of nutrients, and soil types (Jonker et al., 2013; Lamers et al., 2013). In particular, optimizing forest management for mitigation is a complex issue with many uncertainties and still subject to scientific debate. Intensive forest management activities of the early- to mid-twentieth century as well as other factors such as recovery from past overuse, have led to strong forest C sinks in many OECD regions (Pan et al., 2011; Loudermilk et al., 2013; Nabuurs et al., 2013; Erb et al., 2013). However, the capacity of these sinks is being reduced as forests approach saturation (Smith, 2005; Körner, 2006; Guldea et al., 2008; Nabuurs et al., 2013; Sections 11.2.3, 11.3.2). Active forest management, including management for bioenergy, is therefore important for sustaining the strength of the forest carbon sink well into the future (Nabuurs et al., 2007, 2013; Canadell and Raupach, 2008; Ciais et al., 2008), although countries should realize that for some old forest areas, conserving carbon stocks may be preferential, and that the actively managed forests may for some time (decades) act as sources. Organic wastes include waste from households and restaurants, discarded wood products such as paper, construction, and demolition wood waste, and waste waters suitable for anaerobic biogas production (Haberl et al., 2010; Gregg and Smith, 2010). Organic waste may be dispersed and also heterogeneous in quality but the health and environmental gains from collection and proper management through combustion or anaerobic digestion can be significant. Competition with alternative uses of the wastes may limit this resource potential. Dedicated biomass plantations include annual (cereals, oil, and sugar crops) and perennial plants (e.g., switchgrass, Miscanthus) and tree plantations (both coppice and single-stem plantations (e.g., willow, poplar, eucalyptus, pine; (Hoogwijk et al., 2005, 2009; Smeets et al., 2007; van Vuuren et al., 2009; Dornburg et al., 2010; Wicke et al., 2011b; Haberl et al., 2011a). The range of estimates of technical bioenergy potentials from that resource in 2050 is particularly large (500 EJ/yr). Technical bioenergy potentials from dedicated biomass plantations are generally calculated by multiplying (1) the area deemed available for energy crops by (2) the yield per unit area and year (Batidzirai et al., 2012; Coelho et al., 2012). Some studies have identified a sizable technical potential (up to 100 EJ) for bioenergy production using marginal and degraded lands (e.g., saline land) that are currently not in use for food production or grazing (Nijsen et al., 2012). However, how much land is really unused and available is contested (Erb et al., 2007; Haberl et al., 2010; Coelho et al., 2012). Contrasting views on future technical bioenergy potentials from dedicated biomass plantations can be explained by differences in assumptions regarding feasible future agricultural crop yields, livestock feeding efficiency, land availability for energy crops and yields of energy crops (Dornburg et al., 2010; Batidzirai et al., 2012; Erb et al., 2012a). Most scientists agree that increases in food crop yields and higher feeding efficiencies and lower consumption of animal products results in higher technical bioenergy potential. Also, there is a large agreement that careful policies for implementation focused on land-use zoning approaches (including nature conservation and biodiversity protection), multifunctional land use, integration of food and energy production, avoidance of detrimental livelihood impacts, e.g., on livestock grazing and subsistence farming, and consideration of equity issues, and sound management of impacts on water systems are crucial for sustainable solutions. Reduced traditional biomass demand. A substantial quantity of biomass will become available for modern applications by improving the end-use efficiency of traditional biomass consumption for energy, mostly in households but also within small industries (such as charcoal kilns, brick kilns, etc.). Traditional bioenergy represents approximately 15% of total global energy use and 80% of current bioenergy use (≈35 EJ/yr) and helps meeting the cooking needs of ~2.6 billion people (Chum et al., 2011; IEA, 2012b). Traditional bioenergy use covers several end-uses including cooking, water, and space heating, and small-industries (such as brick and pottery kilns, bakeries, and many others). Cooking is the dominant end use; it is mostly done in open fires and rudimentary stoves, with approximately 10–20% conversion efficiency, leading to very high primary energy consumption. Advanced woodburning and biogas stoves can potentially reduce biomass fuel consumption by 60% or more (Jetter et al., 2012) and further lower the atmospheric radiative forcing, reducing CO_2 emissions, and in many cases black carbon emissions, by up to 90% (Anenberg et al., 2013). Assuming that actual savings reach on average 30–60% of current consumption, the total bioenergy potential from reducing traditional bioenergy demand can be estimated at 8–18 EJ/yr. An unknown fraction of global traditional biomass is consumed in a non-environmentally sustainable way, leading to forest degradation and deforestation.

Detailed country studies have estimated the fraction of non-renewable biomass from traditional bioenergy use to vary widely, e.g., from 1.6% for the Democratic Republic of Congo to 73% for Burundi (CDM-SSC WG, 2011) with most countries in the range between 10–30% (i.e., meaning that 70–90% of total traditional bioenergy use is managed sustainably). Thus a fraction of the traditional biomass saved through better technology, should not be used for other energy purposes but simply not consumed to help restore the local ecosystems.

11.13.3 Bioenergy conversion: technologies and management practices

Numerous conversion technologies can transform biomass to heat, power, liquid, and gaseous fuels for use in the residential, industrial, transport, and power sectors (see Chum et al., 2011; GEA, 2012) for a comprehensive coverage of each alternative, and Figure 11.21 for the pathways concerning liquid and gaseous fuels). Since SRREN, the major advances in the large-scale production of bioenergy include the increasing use of hybrid biomass-fossil fuel systems. For example, current commercial coal and biomass co-combustion technologies are the lowest-cost technologies for implementing renewable energy policies, enabled by the large-scale pelletized feedstocks trade (REN21, 2013; Junginger et al., 2014). Direct biopower use is also increasing commercially on a global scale (REN21, 2013, p. 21). In fact, using biomass for electricity and heat, for example, co-firing of woody biomass with coal in the near term and large heating systems coupled with networks for district heating, and biochemical processing of waste biomass, are among the most cost-efficient and effective biomass applications for GHG emission reduction in modern pathways (Sterner and Fritsche, 2011). Integrated gasification combined cycle (IGCC) technologies for coproduction of electricity and liquid fuels from coal and biomass with higher efficiency than current commercial processes are in demonstration phase to reduce cost (Williams et al., 2011; GEA, 2012; Larson et al., 2012). Coupling of biomass and natural gas for fuels is another option for liquid fuels (Baliban et al., 2013) as the biomass gasification technology development progresses. Simulations suggest that integrated gasification facilities are technically feasible (with up to 50% biomass input; Meerman et al., 2011), and economically attractive with a CO2 price of about 66 USD2010/tCO2 (50 EUR2010/tCO2) (Meerman et al., 2012). Many gasification technology developments around the world are in pilot, demonstration, operating first commercial scale for a variety of applications (see examples in Bacovsky et al., 2013; Balan et al., 2013). Many pathways and feedstocks (Figure 11.21) can lead to biofuels for aviation. The development of biofuel standards started and enabled testing of 50% biofuel in jet fuel for commercial domestic and transatlantic flights by consortia of governments, aviation industry, and associations (IEA, 2010; REN21, 2013). Advanced 'drop in' fuels, such as iso-butanol, synthetic aviation kerosene from biomass gasification or upgrading of pyrolysis liquids, can be derived through a number of possible conversion routes such as hydro treatment of vegetable oils, iso-butanol, and Fischer-Tropsch synthesis from gasification of biomass (Hamelinck and Faaij, 2006; Bacovsky et al., 2010; Meerman et al., 2011, 2012; Rosillo-Calle et al., 2012); see also Chapter 8). In specific cases, powering electric cars with electricity from biomass has higher land-use efficiency and lower global-warming potential (GWP) effects than the usage of bioethanol from biofuel crops for road transport across a range of feedstocks, conversion technologies, and vehicle classes (Campbell et al., 2009; Schmidt et al., 2011)13, though costs are likely to remain prohibitive for considerable time (van Vliet et al., 2011a; b; Schmidt et al., 2011). The number of routes from biomass to a broad range of biofuels, shown in Figure 11.21, includes hydrocarbons connecting today's fossil fuels industry in familiar thermal/catalytic routes such as gasification (Williams et al., 2011; Larson et al., 2012) and pyrolysis (Brown et al., 2011; Bridgwater, 2012; Elliott, 2013; Meier et al., 2013). In addition, advances in genomic technology, the emphasis in systems approach, and the integration between engineering, physics, chemistry, and biology bring together many new approaches to biomass conversion (Liao and Messing, 2012) such as (1) biomolecular engineering (Li et al., 2010; Favaro et al., 2012; Peralta-Yahya et al., 2012; Lee et al., 2013; Yoon et al., 2013); (2) deconstruction of lignocellulosic biomass through combinations of mild thermal and biochemical routes in multiple sequential or consolidated steps using similar biomolecular engineering tools (Rubin, 2008; Chundawat et al., 2011; Beckham et al., 2012; Olson et al., 2012; Tracy et al., 2012; Saddler and Kumar, 2013; Kataeva et al., 2013); and (3) advances in (bio)catalysis and basic understanding of the synthesis of cellulose are leading to routes for many fuels and chemicals under mild conditions (SerranoRuiz et al., 2010; Carpita, 2012; Shen et al., 2013; Triantafyllidis et al., 2013; Yoon et al., 2013). Fundamental understanding of biofuel production increased for microbial genomes by forward engineering of cyanobacteria, microalgae, aiming to arrive at minimum genomes for synthesis of biofuels or chemicals (Chen and Blankenship, 2011; Eckert et al., 2012; Ungerer et al., 2012; Jones and Mayfield, 2012; Kontur et al., 2012; Lee et al., 2013). Bioenergy coupled with CCS (Spath and Mann, 2004; Liu et al., 2010) is seen as an option to mitigate climate change through negative emissions if CCS can be successfully deployed (Cao and Caldeira 2010; Lenton and Vaughan 2009). BECCS features prominently in long-run mitigation scenarios (Sections 6.3.2 and 6.3.5) for two reasons: (1) The potential for negative emissions may allow shifting emissions in time; and (2) in scenarios, negative emissions from BECCS compensate for residual emissions in other sectors (most importantly transport) in the second half of the 21st century. As illustrated in Figure 11.22, BECCS is markedly different than fossil CCS because it not only reduces CO2 emissions by storing C in long-term geological sinks, but it continu ally sequesters CO2 from the air through regeneration of the biomass resource feedstock. BECCS deployment is in the development and exploration stages. The most relevant BECCS project is the 'Illinois Basin—Decatur Project' that is projected to inject 1 MtCO2/yr (Gollakota and McDonald, 2012; Senel and Chugunov, 2013). In the United States, two ethanol fuel production by fermentation facilities are currently integrated commercially with carbon dioxide capture, pipeline transport, and use in enhanced oil recovery in nearby facilities at a rate of about 0.2 MtCO2/yr (DiPietro et al., 2012). Altogether, there are 16 global BECCS projects in exploration stage (Karlsson and Byström, 2011). Critical to overall CO2 storage is the realization of a lignocellulosic biomass supply infrastructure for large-scale commodity feedstock production and efficient advanced conversion technologies at scale; both benefit from cost reductions and technological learning as does the integrated system with CCS, with financial and institutional conditions that minimize the risks of investment and facilitate dissemination (Eranki and Dale, 2011; IEA, 2012c, 2013). Integrated analysis is needed to capture system and knock-on effects for bioenergy potentials. A nascent feedstock infrastructure for densified biomass trading globally could indicate decreased pressure on the need for closely co-located storage and production (IEA, 2011; Junginger et al., 2014). The overall technical potential is estimated to be around 10 GtCO2 storage per year for both Integrated Gasification Combined Cycle (IGCC)-CCS co-firing (IGCC with co-gasification of biomass), and Biomass Integrated Gasification Combined Cycle (BIGCC)-CCS dedicated, and around 6 GtCO2 storage for biodiesel based on gasification and Fischer-Tropsch synthesis (FT diesel), and 2.7 GtCO2 for biomethane production (Koornneef et al., 2012, 2013). Another study estimates the potential capacity (similar to technical potential) to be between 2.4 and 10 GtCO2 per year for 2030–2050 (McLaren, 2012). The economic potential, at a CO2 price of around 70 USD/t is estimated to be around 3.3 GtCO2, 3.5 GtCO2, 3.1 GtCO2 and 0.8 GtCO2 in the corresponding four cases, judged to be those with highest economic potential (Koornneef et al., 2012, 2013). Potentials are assessed on a route-by-route basis and cannot simply be added, as they may compete and substitute each other. Practical figures might be not much higher than 2.4 GtCO2 per year at 70–250 USD/tCO2 (McLaren, 2012). Altogether, until 2050, the economic potential is anywhere between 2–10 GtCO2 per year. Some climate stabilization scenarios see considerable higher deployment towards the end of the century, even in some 580–650 ppm scenarios, operating under different time scales, socioeconomic assumptions, technology portfolios, CO2 prices, and interpreting BECCS as part of an overall mitigation framework (e.g., Rose et al., 2012; Kriegler et al., 2013; Tavoni and Socolow, 2013). Possible climate risks of BECCS relate to reduction of land carbon stock, feasible scales of biomass production and increased N2O emis sions, and potential leakage of CO2, which has been stored in deep geologic reservoirs (Rhodes and Keith, 2008). The assumptions of sufficient spatially appropriate CCS capture, pipeline, and storage infrastructure are uncertain. The literature highlights that BECCS as well as CCS deployment is dependent on strong financial incentives, as they are not cost competitive otherwise (Sections 7.5.5; 7.6.4; 7.9; 7.12). Figure 11.22 illustrates some GHG effects associated with BECCS pathways. Tradeoffs between CO2 capture rate and feedstock conversion efficiency are possible. Depicted are pathways with the highest removal rate but not necessarily with the highest feedstock conversion rate. Among all BECCS pathways, those based on integrated gasification combined cycle produce most significant geologic storage potential from biomass, alone (shown in Figure 11.23, electricity) or coupled with coal. Fischer-Tropsch diesel fuel production with biomass as feedstock and CCS attached to plant facilities could enable BECCS for transport; uncertainties in input factors, and output metrics warrant further research (van Vliet et al., 2009). Fischer-Tropsch diesel would also allow net removal but at lower rates than BIGCC. Economics of scale in power plant size are crucial to improve economic viability of envisaged BECCS projects. Increasing power plant size requires higher logistic challenges in delivering biomass. Scales of 4,000 to 10,000 Mg/day needed for > 600 MW power plants could become feasible as the biomass feedstock supply logistic development with manageable logistic costs if biomass is derived from high-yield monocrops; logistical costs are more challenging when biomass is derived from residues (e.g., Argo et al., 2013; Junginger et al., 2014). Large-scale biomass production with flexible integrated polygeneration facilities for fuels and/or power can improve the technoeconomic performance, currently above market prices to become more economically competitive over time (Meerman et al., 2011). In the future, increased operating experience of BECCS IGCC-CCS through technological improvements and learning could enable carbon neutral electricity and, in combination with CCS, could result in net removal of CO2 (Figure 11.22). BECCS is among the lowest cost CCS options for a number of key industrial sectors (Meerman et al., 2013). It should be noted that primary empiric cost and performance data for dedicated bioenergy plants are not yet available and needed for comprehensively assessing BECCS. The current status of CCS and on-going research issues are discussed in Sections 7.5.5 and 7.6.4. Social concerns constitute a major barrier for implement demonstration and deployment projects. Integrated bio-refineries continue to be developed; for instance, 10% of the ethanol or corresponding sugar stream goes into bio-products in Brazil (REN21, 2012) including making ethylene for polymers (IEAETSAP and IRENA, 2013). Multi product bio-refineries could produce a wider variety of co-products to enhance the economics of the overall process, facilitating learning in the new industry (IEA, 2011); Lifecycle Analyses (LCAs) for these systems are complex (Pawelzik et al., 2013).

There are alternatives to land-based bioenergy. Microalgae, for example, offer a high-end technical potential. However, it might be compromised by water supply, if produced in arid land, or by impacts on ocean ecosystems. To make microalgae cost competitive, maximizing algal lipid content (and then maximizing growth rate) requires technological breakthroughs (Davis et al., 2011a; Sun et al., 2011; Jonker and Faaij, 2013). The market potential depends on the co-use of products for food, fodder, higher value products, and fuel markets (Chum et al., 2011). Similarly, lignocellulosic feedstocks produced from waste or residues, or grown on land unsupportive of food production (e.g., contaminated land for remediation as in previously mined land) have been suggested to reduce socio-environmental impact. In addition, lignocellulosic feedstocks can be bred specifically for energy purposes, and can be harvested by coupling collection and pre-processing (densification and others) in depots prior to final conversion, which could enable delivery of more uniform feedstocks throughout the year (Eranki and Dale, 2011; U.S. DOE, 2011; Argo et al., 2013). Various conversion pathways are in research and development (R&D), near commercialization, or in early deployment stages in several countries (see Section 2.6.3 in Chum et al., 2011). More productive land is also more economically attractive for cellulosic feedstocks, in which case competition with food production is more likely. Depending on the feedstock, conversion process, prior land use, and land demand, lignocellulosic bioenergy can be associated with high or low GHG emissions (e.g., Davis et al., 2011b). Improving agricultural lands and reducing non-point pollution emissions to watersheds remediate nitrogen run off and increase overall ecosystems' health (Van Dam et al., 2009a; b; Gopalakrishnan et al., 2012). Also regeneration of saline lands by salt-tolerant tree and grass species can have a large potential on global scale as demonstrated by Wicke et al. (2011). A range of agro-ecological options to improve agricultural practices such as no/low tillage conservation, agroforestry, etc., have potential to increase yields (e.g., in sub-Saharan Africa), while also providing a range of co-benefits such as increased soil organic matter. Such options require a much lower level of investment and inputs and are thus more readily applicable in developing countries, while also holding a low risk of increased GHG emissions (Keating et al., 2013). Substantial progress has also been achieved in the last four years in small-scale bioenergy applications in the areas of technology innovation, impact evaluation and monitoring, and in large-scale implementation programmes. For example, advanced combustion biomass cookstoves, which reduce fuel use by more than 60% and hazardous pollutant as well as short-lived climate pollutants by up to 90%, are now in the last demonstration stages or commercial (Kar et al., 2012; Anenberg et al., 2013). Innovative designs include micro-gasifiers, stoves with thermoelectric generators to improve combustion efficiency and provide electricity to charge LED lamps while cooking, stoves with advanced combustion chamber designs, and multi-use stoves (e.g cooking and water heating for bathing (Ürge-Vorsatz et al., 2012; Anenberg et al., 2013). Biogas stoves, in addition to providing clean combustion, help reduce the health risks associated with the disposal of organic wastes. There has also been a boost in cookstove dissemination efforts ranging from regional (multi-country) initiatives (Wang et al., 2013b) to national, and project-level interventions. In total, more than 200 large-scale cookstove projects are in place worldwide, with several million efficient cookstoves installed each year (Cordes, 2011). A Global Alliance for Clean Cookstoves has been launched that is promoting the adoption of 100 million clean and efficient cookstoves per year by 2030 and several countries have launched National Cookstove Programs in recent years (e.g., Mexico, Peru, Honduras, and others). Many cookstove models are now manufactured in large-scale industrial facilities using state-of-the-art materials and combustion design technology. Significant efforts are also in place to develop international standards and regional stove testing facilities. In addition to providing tangible local health and other sustainable benefits, replacing traditional open fires with efficient biomass cookstoves has a global mitigation potential estimated to be between 0.6 and 2.4 $GtCO_2eq$/yr (Ürge-Vorsatz et al., 2012). Small-scale decentralized biomass power generation systems based on biomass combustion and gasification and biogas production systems have the potential to meet the electricity needs of rural communities in the developing countries. The biomass feedstocks for these small-scale systems could come from residues of crops and forests, wastes from livestock production, and/or from small-scale energy plantations (Faaij, 2006).

11.13.4 GHG emission estimates of bioenergy production systems

The combustion of biomass generates gross GHG emissions roughly equivalent to the combustion of fossil fuels. If bioenergy production is to generate a net reduction in emissions, it must do so by offsetting those emissions through increased net carbon uptake of biota and soils. The appropriate comparison is then between the net biosphere flux in the absence of bioenergy compared to the net biosphere flux in the presence of bioenergy production. Direct and indirect effects need to be considered in calculating these fluxes. Bioenergy systems directly influence local and global climate through (i) GHG emissions from fossil fuels associated with biomass production, harvest, transport, and conversion to secondary energy carriers (von Blottnitz and Curran, 2007; van der Voet et al., 2010); (ii) CO_2 and other GHG emissions from biomass or biofuel combustion (Cherubini et al., 2011); (iii) atmosphere-ecosystem exchanges of CO_2 following land disturbance (Berndes et al., 2013; Haberl, 2013); (iv) climate forcing resulting from emissions of short-lived GHGs like black carbon and other chemically active gases (NO_x, CO, etc.) (Tsao et al., 2012; Jetter et al., 2012); (v) climate forcing resulting from alteration of biophysical properties of the land surface affecting the surface energy balance (e.g., from changes in surface albedo, heat and water fluxes, surface roughness, etc.; (Bonan, 2008; West et al., 2010a; Pielke Sr. et al., 2011); and (vi) GHGs from land management and perturbations to soil biogeochemistry, e.g., N_2O from fertilizers, CH_4, etc. (Cai, 2001; Allen et al., 2009). Indirect effects include the partial or complete substitution of fossil fuels and the indirect transformation of land use by equilibrium effects. Hence, the total climate forcing of bioenergy depends on feedstock, site-specific climate and ecosystems, management conditions, production pathways, end use, and on the interdependencies with energy and land markets. In contrast, bioenergy systems have often been assessed (e.g., in LCA studies, integrated models, policy directives, etc.) under the assumption that the CO_2 emitted from biomass combustion is climate neutral14 because the carbon that was previously sequestered from the atmosphere will be re-sequestered if the bioenergy system is managed sustainably (Chum et al., 2011; Creutzig et al., 2012a; b). The shortcomings of this assumption have been extensively discussed in environmental impact studies and emission accounting mechanisms (Searchinger et al., 2009; Searchinger, 2010; Cherubini et al., 2011; Haberl, 2013). Studies also call for a consistent and case-specific carbon stock/flux change accounting that integrates the biomass system with the global carbon cycle (Mackey et al., 2013). As shown in Chapter 8 of WGI (Myhre and Shindell, 2013) and (Plattner et al., 2009; Fuglestvedt et al., 2010), the climate impacts can be quantified at different points along a cause-effect chain, from emissions to changes in temperature and sea level rise. While a simple sum of the net CO_2 fluxes over time can inform about the skewed time distribution between sources and sinks ('C debt'; Marland and Schlamadinger, 1995; Fargione et al., 2008; Bernier and Paré, 2013), understanding the climate implication as it relates to policy targets (e.g., limiting warming to 2 °C) requires models and/or metrics that also include temperature effects and climate consequence (Smith et al., 2012c; Tanaka et al., 2013). While the warming from fossil fuels is nearly permanent as it persists for thousands of years, direct impacts from renewable bioenergy systems cause a perturbation in global temperature that is temporary and even at times cooling if terrestrial carbon stocks are not depleted (House et al., 2002; Cherubini et al., 2013; Joos et al., 2013; Mackey et al., 2013). The direct, physical climate effects at various end-points need to be fully understood and characterized—despite the measurement challenges that some climate forcing mechanisms can entail (West et al., 2010b; Anderson-Teixeira et al., 2012), and coherently embedded in mitigation policy scenarios along with the possible counterfactual effects. For example, in the specific case of existing forests that may continue to grow if not used for bioenergy, some studies employing counterfactual baselines show that forest bioenergy systems can temporarily have higher cumulative CO_2 emissions than a fossil reference system (for a time period ranging from a few decades up to several centuries; (Repo et al., 2011; Mitchell et al., 2012; Pingoud et al., 2012; Bernier and Paré, 2013; Guest et al., 2013; Helin et al., 2013; Holtsmark, 2013). In some cases, cooling contributions from changes in surface albedo can mitigate or offset these effects (Arora and Montenegro, 2011; O'Halloran et al., 2012; Anderson-Teixeira et al., 2012; Hallgren et al., 2013). Accounting always depends on the time horizon adopted when assessing climate change impacts, and the assumed baseline, and hence includes value judgements (Schwietzke et al., 2011; Cherubini et al., 2013; Kløverpris and Mueller, 2013). Two specific contributions to the climate forcing of bioenergy, not addressed in detail in SRREN include N2O and biogeophysical factors. Nitrous oxide emissions: For first-generation crop-based biofuels, as with food crops (see Chapter 11), emissions of N2O from agricultural soils is the single largest contributor to direct lifecycle GHG emissions, and one of the largest contributors across many biofuel production cycles (Smeets et al., 2009a; Hsu et al., 2010). Emission rates can vary by as much as 700% between different crop types for the same site, fertilization rate, and measurement period (Kaiser and Ruser, 2000; Don et al., 2012; Yang et al., 2012). Increased estimates of N2O emissions alone can convert some biofuel systems from apparent net sinks to net sources (Crutzen et al., 2007; Smith et al., 2012c). Improvements in nitrogen use efficiency and nitrogen inhibitors can substantially reduce emissions of N2O (Robertson and Vitousek, 2009). For some specific crops, such as sugarcane, N2O emissions can be low (Macedo et al., 2008; Seabra et al., 2011) or high (Lisboa et al., 2011). Other bioenergy crops require minimal or zero N fertilization and can reduce GHG emissions relative to the former land use where they replace conventional food crops (Clair et al., 2008).

Biogeophysical factors: Land cover changes or land-use disturbances of the surface energy balance, such as surface albedo, surface roughness, and evapotranspiration influence the climate system (Betts, 2001; Marland et al., 2003; Betts et al., 2007; Bonan, 2008; Jackson et al., 2008; Mahmood et al., 2013). Perturbations to these can lead to both direct and indirect climate forcings whose impacts can differ in spatial extent (global and/or local) (Bala et al., 2007; Davin et al., 2007). Surface albedo is found to be the dominant direct biogeophysical climate impact mechanism linked to land cover change at the global scale, especially in areas with seasonal snow cover (Claussen et al., 2001; Bathiany et al., 2010), with radiative forcing effects possibly stronger than those of the co-occuring C-cycle changes (Randerson et al., 2006; Lohila et al., 2010; Bright et al., 2011; Cherubini et al., 2012; O'Halloran et al., 2012). Land cover changes can also affect other biogeophysical factors like evapotranspiration and surface roughness, which can have important local (Loarie et al., 2011; Georgescu et al., 2011) and global climatic consequences (Bala et al., 2007; Swann et al., 2010, 2011). Biogeophysical climate impacts from changes in land use are site-specific and show variations in magnitude across different geographic regions and biomes (Bonan, 2008; Anderson, 2010; Pielke Sr. et al., 2011; Anderson-Teixeira et al., 2012). Biogeophysical impacts should be considered in climate impact assessments and in the design of land-use policies to adequately assess the net impacts of land-use mitigation options (Jackson et al., 2008; Betts, 2011; Arora and Montenegro, 2011) as their size may be comparable to impacts from changes to the C cycle. Figure 11.23 illustrates the range of lifecycle global direct climate impact (in g CO2 equivalents per MJ, after characterization with GWP time horizon=100 years) attributed to major global bioenergy products reported in the peer-reviewed literature after 2010. Results are broadly comparable to those of Chapter 2 in SRREN (Figures 2.10 and 2.11 in SRREN; Chum et al., 2011) Those figures displayed negative emissions, resulting from crediting emission reduction due to substitution effects. This appendix refrains from allocating credits to feedstocks to avoid double accounting. Significant variation in the results reflects the wide range of conversion technologies and the reported performances in addition to analyst assumptions affecting system boundary completeness, emission inventory completeness, and choice of allocation method (among others). Additional 'site-specific' land-use considerations such as changes in soil organic carbon stocks (ΔSOC), changes in surface albedo (Δalbedo), and the skewed time distribution of terrestrial biogenic CO2 fluxes can either reduce or compound land-use impacts and are presented to exemplify that, for some bioenergy systems, these impacts can be greater in magnitude than lifecycle impacts from feedstock cultivation and bioenergy product conversion. 'Site-specific' land-use considerations are geographically explicit and highly sensitive to background climate conditions, soil properties, biomass yields, and land management regimes. The figure reveals that studies find very different values depending on the boundaries of analysis chosen, site-specific effects, and management methods. Nonetheless, it is clear that fuels from sugarcane, perennial grasses, crop residues, and waste cooking oil are more beneficial than other fuels (LUC emissions can still be relevant, see Figure 11.23). Another important result is that albedo effects and site-specific CO2 fluxes are highly variable for different forest systems and environmental conditions and determine the total climate forcing of bioenergy from forestry. Direct and indirect land-use change: Direct land-use change occurs when bioenergy crops displace other crops or pastures or forests, while iLUC results from bioenergy deployment triggering the conversion to cropland of lands, somewhere on the globe, to replace some portion of the displaced crops (Searchinger et al., 2008; Kløverpris et al., 2008; Delucchi, 2010; Hertel et al., 2010). Direct LUC to establish biomass cropping systems can increase the net GHG emissions, for example, if carbon-rich ecosystems such as wetlands, forests, or natural grasslands are brought into cultivation (Gibbs et al., 2008; UNEP, 2009, p. 2009; Chum et al., 2011). Biospheric C losses associated with LUC from some bioenergy schemes can be, in some cases, more than hundred times larger than the annual GHG savings from the assumed fossil fuel replacement (Gibbs et al., 2008; Chum et al., 2011). Impacts have been shown to be significantly reduced when a dynamic baseline includes future trends in global agricultural land use (Kløverpris and Mueller, 2013). Albeit at lower magnitude, beneficial LUC effects can also be observed, for example, when some semi-perennial crops, perennial grasses or woody plants replace annual crops grown with high fertilizer levels, or where such plants are produced on lands with carbon-poor soils (Tilman et al., 2006; Harper et al., 2010; Sterner and Fritsche, 2011; Sochacki et al., 2012). In particular, Miscanthus improves soil organic carbon reducing overall GHG emissions (Brandão et al., 2011); degraded USA Midwest land for economic agriculture, over a 20-year period, shows successional perennial crops without the initial carbon debt and indirect land-use costs associated with food-based biofuels (Gelfand et al., 2013). Palm oil, when grown on more marginal grasslands, can deliver a good GHG balance and net carbon storage in soil (Wicke et al., 2008). Such lands represent a substantial potential for palm oil expansion in Indonesia without deforestation and draining peat lands (Wicke et al., 2011a). In long-term rotation forests, the increased removal of biomass for bioenergy may be beneficial or not depending on the site-specific forest conditions (Cherubini et al., 2012b). For long-term rotation biomass, the carbon debt (increased cumulative CO2 emissions for a duration in the order of a rotation cycle or longer) becomes increasingly important (Schlamadinger and Marland, 1996; Marland and Schlamadinger, 1997; Fargione et al., 2008; McKechnie et al., 2011; Hudiburg et al., 2011). Calculations of specific GHG emissions from long-term rotation forests need to account for the foregone CO2-accumulation (Searchinger, 2010; Holtsmark, 2012; Pingoud et al., 2012; Haberl et al., 2012).

If part of a larger forest is used as a feedstock for bioenergy while the overall forest carbon stock increases (the so-called landscape perspective), then the overall mitigation effects are positive, in particular over several harvesting cycles making use of the faster carbon sequestration rates of younger forests (Daigneault et al., 2012; Ximenes et al., 2012; Lamers and Junginger, 2013; Latta et al., 2013). Nabuurs et al. (2013) observe first signs of a carbon sink saturation in European forest biomass and suggest to focus less on the forest biomass sink strength but to consider a mitigation strategy that maximizes the sum of all the possible components: (1) carbon sequestration in forest biomass; (2) soil and wood products; and (3) the effects of material and energy substitution of woody biomass. In general, the use of easily decomposable residues and wastes for bioenergy can produce GHG benefits (Zanchi et al., 2012), similarly to increasing the biomass outtake from forests affected by high mortality rates (Lamers et al., 2013), whereas the removal of slowly decomposing residues reduces soil carbon accumulation at a site and results in net emissions (Repo et al., 2011). The anticipation of future bioenergy markets may promote optimized forest management practices or afforestation of marginal land areas to establish managed plantations, thus contributing to increased forest carbon stocks (Sedjo and Tian, 2012). Rather than leading to wide-scale loss of forest lands, growing markets for tree products can provide incentives for maintaining or increasing forest stocks and land covers, and improving forest health through management (Eisenbies et al., 2009; Dale et al., 2013). If managed to maximize CO2 storage rate over the long-term, long-term rotation forests offer low-cost mitigation options, in particular, when woody products keep carbon within the human-built environment over long time-scales (e.g., wood substituting for steel joist; (Lippke et al., 2011). Indirect land-use change is difficult to ascertain because the magnitude of these effects must be modelled (Nassar et al., 2011) raising important questions about model validity and uncertainty (Liska and Perrin, 2009; Plevin et al., 2010; Khanna et al., 2011; Gawel and Ludwig, 2011; Wicke et al., 2012) and policy implications (DeCicco, 2013; Finkbeiner, 2013; Plevin et al., 2013). Available model-based studies have consistently found positive and, in some cases, high emissions from LUC and iLUC, mostly of first-generation biofuels (Figure 11.23), albeit with high variability and uncertainty in results (Hertel et al., 2010; Taheripour et al., 2011; Dumortier et al., 2011; Havlík et al., 2011; Chen et al., 2012; Timilsina et al., 2012; Warner et al., 2014). Causes of the great uncertainty include: incomplete knowledge on global economic dynamics (trade patterns, land-use productivity, diets, use of by-products, fuel prices, and elasticities); selection of specific policies modelled; and the treatment of emissions over time (O'Hare et al., 2009; Khanna et al., 2011; Wicke et al., 2012). In addition, LUC modelling philosophies and model structures and features (e.g., dynamic vs. static model) differ among studies. Variations in estimated GHG emissions from biofuel-induced LUC are also driven by differences in scenarios assessed, varying assumptions, inconsistent definitions across models (e.g., LUC, land type), specific selection of reference sce narios against which (marginal) LUC is quantified, and disparities in data availability and quality. The general lack of thorough sensitivity and uncertainty analysis hampers the evaluation of plausible ranges of estimates of GHG emissions from LUC. Wicke et al. (2012) identified the need to incorporate the impacts of iLUC prevention or mitigation strategies in future modelling efforts, including the impact of zoning and protection of carbon stocks, selective sourcing from low risk-areas, policies and investments to improve agricultural productivity, double cropping, agroforestry schemes, and the (improved) use of degraded and marginal lands (see Box 7.1). Indirect land-use change is mostly avoided in the modelled mitigation pathways in Chapter 6. The relatively limited fuel coverage in the literature precludes a complete set of direct comparisons across alternative and conventional fuels sought by regulatory bodies and researchers. GHG emissions from LUC can be reduced, for instance through production of bioenergy co-products that displace additional feedstock requirements, thus decreasing the net area needed (e.g., for corn, Wang et al., 2011a; for wheat, Berndes et al., 2011). Proper management of livestock and agriculture can lead to improved resource efficiency, lower GHG emissions, and lower land use while releasing land for bioenergy production as demonstrated for Europe (de Wit et al., 2013) and Mozambique (van der Hilst et al., 2012b). For land transport, cellulosic biomass, such as Miscanthus, has been suggested as a relatively low-carbon source for bioethanol that could be produced at scale, but only if iLUC can be avoided by not displacing food and other commodities and if comprehensive national land management strategies are developed (e.g., Dornburg et al., 2010; Scown et al., 2012). Negative iLUC values are theoretically possible (RFA, 2008). Producing biofuels from wastes and sustainably harvested residues, and replacing first-generation biofuel feedstocks with lignocellulosic crops (e.g., grasses) would induce little or no iLUC (Davis et al., 2011b; Scown et al., 2012). While iLUC quantifications remain uncertain, lower agricultural yields, land-intensive diets, and livestock feeding efficiencies, stronger climate impacts and higher energy crop production levels can result in higher LUC-related GHG emissions. Strong global and regional governance (forest protection, zoning), technological change in agriculture and biobased options, and high-yield bioenergy crops and use of residues and degraded land (if available) could all reduce iLUC (Van Dam et al., 2009a; b; Wicke et al., 2009; Fischer et al., 2010; de Wit et al., 2011, 2013; van der Hilst et al., 2012a; Rose et al., 2013).

As with any other renewable fuel, bioenergy can replace or complement fossil fuel. The fossil fuel replacement effect, relevant when a global cap on CO2 emissions is absent, is discussed in Chapter 8.7. Indirect effects are not restricted to indirect GHG effects of production of biomass in agricultural systems; there are also indirect (market-mediated) effects of wood energy, but also effects in terms of biodiversity threats, environmental degradation, and external social costs, which are not considered here.

11.13.5 Aggregate future potential deployment in integrated models

In SRREN scenarios (IPCC, 2011), bioenergy is projected to contribute 80–190 EJ/yr to global primary energy supply by 2050 for 50% of the scenarios in the two mitigation levels modelled. The min to max ranges were 20–265 EJ/yr for the less stringent scenarios and 25–300 EJ for the tight mitigation scenarios (The WGIII AR5 transformation pathway studies suggest that modern bioenergy could play a significant role within the energy system (Section 6.3.5) providing 5 to 95 EJ/yr in 2030, 10 to 245 EJ/yr in 2050, and 105 to 325 EJ/yr in 2100 under idealized full implementation scenarios (see also Figure 7.12), with immediate, global, and comprehensive incentives for land-related mitigation options. The scenarios project increasing deployment of bioenergy with tighter climate change targets, both in a given year as well as earlier in time (see Figure 6.20). Models project increased dependence, as well as increased deployment, of modern bioenergy, with some models projecting 35% of total primary energy from bioenergy in 2050, and as much as 50% of total primary energy from modern bioenergy in 2100. Bioenergy's share of regional total electricity and liquid fuels could be significant—up to 35% of global regional electricity from biopower by 2050, and up to 70% of global regional liquid fuels from biofuels by 2050. However, the cost-effective allocation of bioenergy within the energy system varies across models. Several sectoral studies, focusing on biophysical constraints, model assumptions (e.g., estimated increase in crop yields over large areas) and current observations, suggest to focus on the lower half of the ranges reported above (Field et al., 2008; Campbell et al., 2008; Johnston et al., 2009a, 2011; Haberl et al., 2013b).

BECCS features prominently in many mitigation scenarios. BECCS is deployed in greater quantities and earlier in time the more stringent the climate policy (Section 6.3.5). Whether BECCS is essential for mitigation, or even sufficient, is unclear. In addition, the likelihood of BECCS deployment is difficult to evaluate and depends on safety confirmations, affordability and public acceptance (see Section 11.13.3 for details).

BECCS may also affect the cost-effective emissions trajectory (Richels et al., In Review; Rose et al., 2013). Some integrated models are cost-effectively trading off lower land carbon stocks and increased land N2O emissions for the long-run mitigation benefits of bioenergy (Rose et al., 2013; Popp et al., 2013). The models find that bioenergy could contribute effectively to climate change mitigation despite land conversion and intensification emissions. However, as discussed below and in Section 11.9, policy implementation and coordination are factors to consider. In these models, constraining bioenergy has a cost. For instance, limiting global bioenergy availability to 100 EJ/year tripled marginal abatement costs and doubled consumption losses associated with transformation pathways (Rose et al., 2013). Overall outcomes may depend strongly on governance of land use and deployment of best practices in agricultural production (see sections above). Progressive developments in governance of land and modernization of agriculture and livestock and effective sustainability frameworks can help realize large parts of the technical bioenergy potential with low associated GHG emissions. With increasing scarcity of productive land, the growing demand for food and bioenergy could induce substantial LUC causing high GHG emissions and/or increased agricultural intensification and higher N2O emissions unless wise integration of bioenergy into agriculture and forestry landscapes occurs (Delucchi, 2010). Consideration of LUC emissions in integrated models show that valuing or protecting global terrestrial carbon stocks reduces the potential LUC-related GHG emissions of energy crop deployment, and could lower the cost of achieving climate change objectives, but could exacerbate increases in agricultural commodity prices (Popp et al., 2011; Reilly et al., 2012). Many integrated models are investigating idealized policy implementation pathways, assuming global prices on GHG (including the terrestrial land carbon stock); if such conditions cannot be realized, certain types of bioenergy could lead to additional GHG emissions. More specifically, if the global terrestrial land carbon stock remains unprotected, large GHG emissions from bioenergy-related LUC alone are possible (Melillo et al., 2009; Wise et al., 2009; Creutzig et al., 2012a; Calvin et al., 2013b). In summary, recent integrated model scenarios project between 10–245 EJ/yr modern bioenergy deployment in 2050. Good governance and favourable conditions for bioenergy development may facilitate higher bioenergy deployment while sustainability and livelihood concerns might constrain deployment of bioenergy scenarios to low values (see Section 11.13.6).

11.13.6 Bioenergy and sustainable development

The nature and extent of the impacts of implementing bioenergy depend on the specific system, the development context, and on the size of the intervention (Section 11.4.5). The effects on livelihoods have not yet been systematically evaluated in integrated models (Davis et al., 2013; Creutzig et al., 2012b; Creutzig et al., 2013; Muys et al., 2014), even if human geography studies have shown that bioenergy deployment can have strong distributional impacts (Davis et al., 2013; Muys et al., 2014). The total effects on livelihoods will be mediated by global market dynamics, including policy regulations and incentives, the production model and deployment scale, and place-specific factors such as governance, land tenure security, labour and financial capabilities, among others (Creutzig et al., 2013). Bioenergy projects can be economically beneficial, e.g., by raising and diversifying farm incomes and increasing rural employment through the production of biofuels for domestic use (Gohin, 2008) or export markets (Wicke et al., 2009; Arndt et al., 2011). The establishment of large-scale biofuels feedstock production can also cause smallholders, tenants, and herders to lose access to productive land, while other social groups such as workers, investors, company owners, biofuels consumers, and populations who are more responsible for GHG emission reductions enjoy the benefits of this production (van der Horst and Vermeylen, 2011). This is particularly relevant where large areas of land are still unregistered or are being claimed and under dispute by several users and ethnic groups (Dauvergne and Neville, 2010). Furthermore, increasing demand for first-generation biofuels is partly driving the expansion of crops like soy and oil palm, which in turn contribute to promote large-scale agribusinesses at the expense of family and community-based agriculture, in some cases (Wilkinson and Herrera, 2010). Biofuels deployment can also translate into reductions of time invested in on-farm subsistence and community-based activities, thus translating into lower productivity rates of subsistence crops and an increase in intra-community conflicts as a result of the uneven share of collective responsibilities (Mingorría et al., 2010). Bioenergy deployment is more beneficial when it is not an additional land-use activity expanding over the landscape, but rather integrates into existing land uses and influences the way farmers and forest owners use their land. Various studies indicate the ecosystem services and values that perennial crops have in restoring degraded lands, via agroforestry systems, controlling erosion, and even in regional climate effects such as improved water retention and precipitation (Faaij, 2006; Wicke et al., 2011c; Immerzeel et al., 2013). Examples include adjustments in agriculture practices where farmers, for instance, change their manure treatment to produce biogas, reduce methane and N losses. Changes in management practice may swing the net GHG balance of options and also have clear sustainable development implications (Davis et al., 2013). Small-scale bioenergy options can provide cost-effective alternatives for mitigating climate change, at the same time helping advance sustainable development priorities, particularly in rural areas of developing countries. IEA (2012b) estimates that 2.6 billion people world wide depend on traditional biomass for cooking, while 84% of these belong to rural communities. Use of low-quality fuels and inefficient cooking and heating devices leads to pollution resulting in nearly 4 million premature deaths every year, and a range of chronic illnesses and other health problems (Lim et al., 2012; see Section 9.7.3.1). Modern small-scale bioenergy technologies such as advanced/efficient cook stoves, biogas for cooking and village electrification, biomass gasifiers, and bagasse-based co-generation systems for decentralized power generation, can provide energy for rural communities with energy services that also promote rural development (IEA, 2011). Such bioenergy systems reduce CO2 emissions from unsustainable biomass harvesting and short-lived climate pollutants, e.g., black carbon, from cleaner combustion (Chung et al., 2012). Scaling up clean cookstove initiatives could not only save 2 million lives a year, but also significantly reduce GHG emissions (Section 11.13.3). Efficient biomass cook stoves and biogas stoves at the same time provide multiple benefits: They reduce the pressure on forests and biodiversity; they reduce exposure to smoke-related health hazards; they reduce drudgery for women in collection fuelwood; and they save money if fuel needs to be purchased (Martin et al., 2011). Benefits from the dissemination of improved cookstoves outweigh their costs by sevenfold, when their health, economic, and environmental benefits are accounted for (Garcia-Frapolli et al., 2010). Table 11.12 presents the implications of bioenergy options in the light of social, institutional, environmental, economic, and technological conditions. The relationship between bioenergy and these conditions is complex and there could be negative or positive implications, depending on the type of bioenergy option, the scale of the production system and the local context. While biofuels can allow the reduction of fossil fuel use and of GHG emissions, they often shift environmental burdens towards land use-related impacts (i.e., eutrophication, acidification, water depletion, ecotoxicity; EMPA, 2012; Smith and Torn, 2013; Tavoni and Socolow, 2013). Co-benefits and adverse side-effects do not necessarily overlap, neither geographically nor socially (Dauvergne and Neville, 2010; Wilkinson and Herrera, 2010; van der Horst and Vermeylen, 2011). The main potential co-benefits are related to access to energy and impacts on the economy and well-being, jobs creation, and improvement of local resilience (Walter et al., 2011; Creutzig et al., 2013). Main risks of crop-based bioenergy for sustainable develop ment and livelihoods include competition for arable land (Haberl et al., 2013b) and consequent impact on food security, tenure arrangements, displacement of communities and economic activities, creation of a driver of deforestation, impacts on biodiversity, water, and soil, or increment in vulnerability to climate change, and unequal distribution of benefits (Sala et al., 2000; Hall et al., 2009; German et al., 2011; Thompson et al., 2011b; IPCC, 2012). Good governance is an essential component of a sustainable energy system. Integrated studies that compare impacts of bioenergy production between different crops and land management strategies show that the overall impact (both ecological and socio-economic) depends strongly on the governance of land use and design of the bioenergy system see van der Hilst et al. (2012) in the European context, and Van Dam et al. (2009a; b) for different crops and scenarios in Argentina). Van Eijck et al. (2012) show similar differences in impacts between the production and use of Jatropha based on smallholder production versus plantation models. This implies that governance and planning have a strong impact on the ultimate result and impact of large-scale bioenergy deployment. Legislation and regulation of bioenergy as well as voluntary certification schemes are required to guide bioenergy production system deployment so that the resources and feedstocks be put to best use, and that (positive and negative) socioeconomic and environmental issues are addressed as production grows (van Dam et al., 2010). There are different options, from voluntary to legal and global agreements, to improve governance of biomass markets and land use that still require much further attention (Verdonk et al., 2007). The integration of bioenergy systems into agriculture and forest landscapes can improve land and water use efficiency and help address concerns about environmental impacts of present land use (Berndes et al., 2004, 2008; Börjesson and Berndes, 2006; Sparovek et al., 2007; Gopalakrishnan et al., 2009, 2011a; b, 2012; Dimitriou et al., 2009, 2011; Dornburg et al., 2010; Batidzirai et al., 2012; Parish et al., 2012; Baum et al., 2012; Busch, 2012), but the global potentials of such systems are difficult to determine (Berndes and Börjesson, 2007; Dale and Kline, 2013). Similarly, existing and emerging guiding principles and governance systems influence biomass resources availability (Stupak et al., 2011). Certification approaches can be useful, but they should be accompanied by effective territorial policy frameworks (Hunsberger et al., 2012).

11.13.7 Tradeoffs and synergies with land, water, food, and biodiversity

This section summarizes results from integrated models (models that have a global aggregate view, but cannot disaggregate place-specific effects in biodiversity and livelihoods discussed above) on land, water, food, and biodiversity. In these models, at any level of future bioenergy supply, land demand for bioenergy depends on (1) the share of bioenergy derived from wastes and residues (Rogner et al., 2012); (2) the extent to which bioenergy production can be integrated with food or fiber production, which ideally results in synergies (Garg et al., 2011; Sochacki et al., 2013) or at least mitigates land-use competition (Berndes et al., 2013); (3) the extent to which bioenergy can be grown on areas with little current or future production, taking into account growing land demand for food (Nijsen et al., 2012); and (4) the volume of dedicated energy crops and their yields (Haberl et al., 2010; Batidzirai et al., 2012; Smith et al., 2012d). Energy crop yields per unit area may differ by factors of > 10 depending on differences in natural fertility (soils, climate), energy crop plants, previous land use, management and technology (Johnston et al., 2009a; Lal, 2010; Beringer et al., 2011; Pacca and Moreira, 2011; Smith et al., 2012a; Erb et al., 2012a). Assumptions on energy crop yields are one of the main reasons for the large differences in estimates of future area demand of energy crops (Popp et al., 2013). Likewise, assumptions on yields, strategies, and governance on future food/feed crops have large implications for assessments of the degree of land competition between biofuels and these land uses (Batidzirai et al., 2012; de Wit et al., 2013). However, across models, there are very different potential landscape transformation visions in all regions (Sections 6.3.5 and 11.9.). Overall, it is difficult to generalize on regional land cover effects of mitigation. Some models assume significant land conversion while other models do not. In idealized implementation scenarios, there is expansion of energy cropland and forest land in many regions, with some models exhibiting very strong forest land expansion and others very little by 2030. Land conversion is increased in the 450 ppm scenarios compared to the 550 ppm scenarios, but at a declining share, a result consistent with a declining land-related mitigation rate with policy stringency. The results of these integrated model studies need to be interpreted with caution, as not all GHG emissions and biogeophysical or socioeconomic effects of bioenergy deployment are incorporated into these models, and as not all relevant technologies are represented (e.g., cascade utilization). Large-scale bioenergy production from dedicated crops may affect water availability and quality (see Section 6.6.2.6), which are highly dependent on (1) type and quantity of local freshwater resources; (2) necessary water quality; (3) competition for multiple uses (agricultural, urban, industrial, power generation), and (4) efficiency in all sector end uses (Gerbens-Leenes et al., 2009; Coelho et al., 2012). In many regions, additional irrigation of energy crops could further intensify existing pressures on water resources (Popp et al., 2011). Studies indicate that an exclusion of severe water scarce areas for bioenergy production (mainly to be found in the Middle East, parts of Asia, and western United States) would reduce global technical bioenergy potentials by 17% until 2050 (van Vuuren et al., 2009). A model comparison study with five global economic models shows that the aggregate food price effect of large-scale lignocellulosic bioenergy deployment (i.e., 100 EJ globally by the year 2050) is significantly lower (+5% on average across models) than the potential price effects induced by climate impacts on crop yields (+25% on average across models (Lotze-Campen et al., 2013). Possibly hence, ambitious climate change mitigation need not drive up global food prices much, if the extra land required for bioenergy production is accessible or if the feedstock, e.g., from forests, does not directly compete for agricultural land. Effective land-use planning and strict adherence to sustainability criteria need to be integrated into large-scale bioenergy projects to minimize competitions for water (for example, by excluding the establishment of biofuel projects in irrigated areas). If bioenergy is not managed properly, additional land demand and associated LUC may put pressures on biodiversity (Groom et al., 2008;

see Section 6.6.2.5). However, implementing appropriate management, such as establishing bioenergy crops in degraded areas represents an opportunity where bioenergy can be used to achieve positive environmental outcomes (Nijsen et al., 2012).