Bioenergy production on degraded and marginal land

Assessing its potentials, economic performance, and environmental impacts for different settings and geographical scales

Bioenergy production on degraded and marginal land: Assessing its potentials, economic performance, and environmental impacts for different settings and geographical scales

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Bioenergy production on degraded and marginal land
Assessing its potentials, economic performance, and environmental impacts for different settings and geographical scales

Bio-energieproductie op gedegradeerd en marginaal land
Evaluatie van de potenties, economische prestaties en milieu-effecten voor verschillende contexten en geografische schaalniveaus

(met een samenvatting in het Nederlands)

Bioenergieproduktion auf degradierten und marginalen Flächen
Beurteilung der Potenziale, Wirtschaftlichkeit und ökologischen Auswirkungen für unterschiedliche Rahmenbedingungen und geographische Ebenen

(mit einer Zusammenfassung auf Deutsch)

Proefschrift

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Für Joseph

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Abbreviations and Units

AE	Alcohol ethoxylates
AGB	Aboveground biomass
ASSOD	Assessment of the Status of Human-Induced Soil Degradation in South and Southeast Asia
BF	Burkina Faso
BGB	Belowground biomass
Bot	Botswana
C	Carbon
CGPM	Crop and grass production model
CH_4	Methane
CO_2	Carbon dioxide
CO_2-eq.	CO_2 equivalent
CNO	Coconut oil
CPO	Crude palm oil
dm	Dry matter
DOM	Dead organic matter
dS	Decisiemens
DSMW	Digital Soil Map of the World
ECe	Electrical conductivity of the saturated soil extract
EFB	Empty fruit bunches
EJ	Exajoule; 1 EJ = 10^{18} Joule
ESP	Exchangeable sodium percentage
FFB	Fresh fruit bunches
Gha	Giga hectare, 1 Gha = 10^{9} hectare
GHG	Greenhouse gas
GJ	Gigajoule; 1 GJ = 10^{9} Joule
GLASOD	Global Assessment of Human-Induced Soil Degradation
GWP	Global warming potential
H	Hour
HWSD	Harmonized World Soil Database
IPOB	Indonesian Palm Oil Board
IPOC	Indonesian Palm Oil Commission
K	Potassium

Ken	Kenya
km	Kilometer
kWh	Kilowatt hour
K_2O	Potassium oxide
l	Litre
LUC	Land use change
Mal	Mali
MF	Management factor
Mha	Million hectare, 1 Mha = 10^6 hectare
MJ	Megajoule, 1 MJ = 10^6 Joule
MPOB	Malaysian Palm Oil Board
m^3	Cubic meter
N	Nitrogen
NaOH	Sodium hydroxide (also known as lye and caustic soda)
NL	The Netherlands
N_2O	Nitrous oxide
OER	Oil extraction rate
P	Phosphorus
PFAD	Palm fatty acid distillate
PJ	Petajoule; 1 PJ = 10^{15} Joule
PKE	Palm kernel expeller
PKO	Palm kernel oil
PKS	Palm kernel shells
PME	Palm methyl ester
POME	Palm oil mill effluent
P_2O_5	Phosphorus pentoxide
RBD	Refined, deodorized and bleached
REDD	Reduced emissions from deforestation and degradation
RSPO	Roundtable for Sustainable Palm Oil
SA	South Africa
Sen	Senegal
t	Metric tonne
Tan	Tanzania
y	Year
Zam	Zambia

Chapter 1

Introduction

1.1 BIOENERGY AND ITS CONTRIBUTION TO A MORE SUSTAINABLE ENERGY SYSTEM

Current global energy supply is primarily based on fossil fuels; coal, mineral oil, and natural gas comprised approximately 80% of global primary energy supply in 2008 (IEA, 2010). This fossil fuel-based energy supply system is widely considered to be unsustainable for a number of reasons. For example, the combustion of fossil fuels is the largest contributor of greenhouse gas (GHG) emissions to the atmosphere and the primary driver of human-induced global climate change (IPCC, 2007). In addition, the unequal geographic distribution of fossil fuels (IEA, 2010) leads to economic and possibly political dependence on a few countries and geopolitical conflict. Moreover, although increasing slowly over time, fossil fuels are fundamentally finite resources (Rogner, 2000). Finally, in addition to the problems linked to its heavy dependence on fossil fuels, the current global energy supply system is unsustainable because it fails to provide billions of people with access to modern energy services (OECD/IEA, 2010b; UNDP, 2010).

Bioenergy is considered one important option in making the future global energy system more sustainable. To begin with, bioenergy has a substantial growth potential and can therefore make a significant contribution to future energy supply (OECD/IEA, 2007; OECD/IEA, 2010b). Secondly, if produced sustainably, bioenergy can reduce GHG emissions compared to fossil fuels (Dornburg *et al.*, 2008). Thirdly, bioenergy is a versatile energy source usable for producing heat and electricity, as well as solid, liquid, and gaseous fuels (Turkenburg, 2000). Fourthly, bioenergy resources are more evenly distributed around the world, decreasing the dependency of energy imports from a small number of countries and increasing local production of energy.

Currently, combustible renewables and waste account for approximately 10% of global energy consumption (IEA, 2010). Bioenergy is the primary source of energy for approximately 2.7 billion people around the globe, and it plays a vital role in meeting local energy demand in many developing countries (OECD/IEA, 2010b). The bulk of bioenergy consumed is traditional biomass such as fuelwood, charcoal, agricultural residues and animal waste (OECD/IEA, 2007). The traditional use of biomass in combination with often-inefficient stoves has many disadvantages including the significant amount of time spent, mainly by women and children, on fuelwood collection; indoor air pollution and deforestation and soil degradation, which is a particular problem for charcoal production in areas surrounding major cities (Karekezi, 2002; OECD/IEA, 2010b). In addition, the dependence on traditional biomass and the lack of modern energy carriers appears to be linked directly to poverty; as income levels decrease, more traditional biomass is consumed by a larger amount of the population (Karekezi, 2002; Amigun *et al.*, 2008; OECD/IEA, 2010b). As a result, solutions to the problems of traditional biomass must also account for the underlying problem of poverty.

Sustainable bioenergy production – whether in the form of modern energy carriers such as transport fuels or electricity, or in traditional energy forms such as fuelwood and their more efficient use – can reduce energy poverty, contribute to rural development and avoid the negative impacts discussed above. In addition, bioenergy production can generate employment and additional income and, thereby, reduce poverty (Bekunda *et al.*, 2009). Other important benefits of sustainable bioenergy production include the diversification of agricultural markets, increasing local production of energy and reducing dependence on costly, imported fuels (Bekunda *et al.*, 2009).

But increasing global trade and consumption of bioenergy in industrialised countries has been accompanied by a growing concern about the environmental, ecological and social impacts of (modern) bioenergy production. This concern has been spurred by reports about bioenergy crop production causing deforestation and the associated loss of biodiversity, greenhouse gas (GHG) emissions from land use change, displacement of forest people and related land conflicts, and rising food commodity prices, to name just a few (Patzek *et al.*, 2005; CREM *et al.*, 2006; Ziegler, 2007; Fargione *et al.*, 2008; Mitchell, 2008; Searchinger *et al.*, 2008). For example, southeast Asian palm oil has been associated with major problems such as clear-cutting of natural rainforest, destruction of ecologically valuable peatland and instigation of social conflicts, and its sustainability has been intensely debated in many countries (Wakker, 2004; Colchester *et al.*, 2006; CREM *et al.*, 2006; Helms *et al.*, 2006).

1.2 THE USE OF DEGRADED AND MARGINAL LAND FOR SUSTAINABLE BIOMASS PRODUCTION

Many of these unintended and undesired effects of bioenergy production are linked to direct land use change (LUC) (conversion of one type of land to another) and indirect LUC (change in land use in one place induced by the expansion of bioenergy production in another place). However, producing bioenergy on degraded or marginal land may avoid negative effects related to land use change (see for example, Fargione *et al.* (2008), Gallagher (2008), Öko-Institute *et al.* (2008), Schubert *et al.* (2009)) because these types of land are largely unsuitable and often economically unattractive for agricultural crop production. There is little or no other use for such land and thus no (in)direct competition with food production or other uses (Gallagher, 2008). Perennial bioenergy production on degraded and marginal land can also sequester carbon, improve soil fertility, and reduce other soil degradation processes such as soil erosion, dispersion, and leaching as a result of above and belowground biomass growth (Lal, 2001; Berndes, 2002; Lal, 2004b; Gibbs *et al.*, 2008; Schubert *et al.*, 2009). Moreover, perennial bioenergy crops cultivated on degraded and marginal land can increase the quantity and variability of biodiversity, especially if monoculture and large fields are avoided and a mixture of ground-covering species are planted (CBD, 2008; Schubert *et al.*, 2009). In addition, producing bioenergy from degraded and marginal land can contribute to rural social and economic development by using land with no or little previous productivity. Based on these presumed positive impacts, some of the initiatives to develop criteria and certification systems for sustainable bioenergy production and trade promote the use of degraded and marginal land (for an overview of ongoing initiatives in biomass and bioenergy certification until the end of 2009 see van Dam *et al.* (2010)). Examples are the European Commission - Renewable Energy Directive (EC-RED) giving a greenhouse gas (GHG) emission credit of 29 g CO_2-eq. GJ^{-1} for biofuels produced on severely degraded and heavily contaminated land (European Commission, 2009), and the Roundtable on Sustainable Biofuels (RSB) (RSB, 2010) and the Roundtable for Sustainable Palm Oil (RSPO) (Roundtable for Sustainable Palm Oil, 2007) encouraging the use of degraded and idle land for biofuels/palm oil production.

Despite its potential advantages, the use of degraded and marginal land for bioenergy production has drawbacks that may limit its economic attractiveness and may affect its sustainability. The most important challenges are related to 1) difficult growing conditions requiring a large degree of effort over a potentially long period of time and that still often leads to lower productivity than high quality land, and 2) degraded land often being a crucial resource for poor rural communities, particularly those with no formal land rights. While these challenges are acknowledged in many studies (*e.g.* Öko-Institute *et al.*, 2009; Schubert *et al.*, 2009; Dornburg *et al.*, 2010; van Dam *et al.*, 2010), little is actually known about the implications for the technical and economic potential, the economic performance, and the environmental impact of bioenergy production on degraded land. Sections 1.2.1 to 1.2.3 provide an overview of the existing literature, its findings, and its limitations with respect to these implications.

1.2.1 Bioenergy production potential from degraded and marginal land

Several studies (Hoogwijk *et al.*, 2003; Hoogwijk *et al.*, 2005; Tilman *et al.*, 2006; Campbell *et al.*, 2008; Dornburg *et al.*, 2010; Nijsen *et al.*, submitted) have investigated the global technical bioenergy potential from degraded land (Table 1.1). The potential is estimated to range between 8 and 147 EJ y^{-1}. The bioenergy potential from degraded and marginal land depends mainly on the available land area and the biomass yields. Regarding the available degraded and marginal land area, it is important to assess 1) when land should be considered degraded or marginal and 2) when it can actually be considered available for bioenergy production. With respect to the former, there are many different definitions for degraded and marginal land. In its broadest sense, degraded and marginal land refers to land with limited usefulness for any production or regulation function (Schubert *et al.*, 2009). But in the discussion about bioenergy production on degraded and marginal land, the term “degraded land” is often used in combination or even synonymously with the terms “marginal land”, “unproductive land”, “low-productive”, “idle land”, “wasteland”, “fallow land” despite their slightly different meanings (Wiegmann *et al.*, 2008; Schubert *et al.*, 2009). Box 1 (adapted from Wiegmann *et al.*, 2008) provides definitions of degraded land, marginal land and other terms associated with degraded and marginal land as applied in this thesis and illustrates their relationships with each other.

In addition to the different types of land associated with degraded and marginal land, there are many different types and causes of degradation (*e.g.* forest degradation as a result of logging, degradation of grassland as a result of grazing, and salinization of land as a result of inappropriate water management), various severity levels (from slightly to extremely severe), many different uses of the degraded and marginal land (*e.g.* unused, intensive and extensive agricultural use), and variation in vegetation cover of degraded and marginal land (from sparse to still quite dense vegetation in logged forests). All of these aspects are important in identifying where degraded and marginal land is located, whether it is suitable and available for bioenergy production, and whether its use for bioenergy production is sustainable. However, the vague definitions of degraded and marginal land given in the literature make it difficult to identify the location of degraded and marginal land in practice. As a result, bioenergy production even on what may be termed “degraded land” and “marginal land” by some can be associated with such unsustainable practices as deforestation, displacement of vulnerable communities, or biodiversity losses. Thus, firstly, the definition of degraded and marginal land must be clarified by establishing a methodology and criteria for identifying degraded land that can genuinely be considered available for sustainable bioenergy production. Secondly, case studies on identifying degraded and marginal land available for bioenergy production must be conducted.

The second aspect in determining the actual availability of degraded and marginal land for bioenergy production is its current use and functions. A fundamental assumption underlying the proposition of using degraded and marginal land (rather than other land types) for bioenergy production is that degraded and marginal land is either unused or put to very little use. If this is the case, then bioenergy production can have positive social impacts by producing fuel and/or fodder from formerly unused and low-productive areas and by improving the quality of the land so that, in the long term, it can be put to other uses (Schubert *et al.*, 2009). However, the assumption that degraded and marginal land is unused does not necessarily hold because degraded and marginal land is in fact often used both intensively (*e.g.* for food production) and extensively (*e.g.* for livestock grazing and fuelwood collection), and it can be a crucial resource for poor rural communities, particularly those with no formal land rights (Berndes, 2002; Gallagher, 2008; Schubert *et al.*, 2009). In addition, degraded and marginal land may still provide some ecosystem functions and support biodiversity levels similar to managed landscapes (Plieninger and Gaertner, 2011). When investigating potentials of bioenergy production on degraded and marginal land, it is therefore essential to investigate its current use and functions.

Box 1: Definitions of degraded land and other types of land associated with degraded land (adapted from Wiegmann *et al.* (2008))

Abandoned agricultural land is land that was previously used for agricultural crop production or as pasture but that has been abandoned and not converted to forest or urban areas (Wiegmann *et al.*, 2008 citing Field *et al.,* 2008).

Degraded land is land that has experienced the long-term loss of ecosystem function and services caused by disturbances from which the system cannot recover unaided (UNEP, 2007).

Fallow land is land on which cultivation has been temporarily suspended for one or more vegetation periods to allow recovery of soil fertility.

Low/high-productive land refers to a spectrum on which land gradually changes from low to high productivity for agriculture and forestry.

Marginal land is land on which cost-effective food and feed production is not possible under given site conditions and cultivation techniques.

Used/unused land refers to a spectrum on which land gradually changes from intensely used land towards land that is not influenced by any anthropogenic land use forms. Unused land refers to both areas of undisturbed wildlife and to abandoned land where former land use activities were discontinued.

Wasteland is characterized by natural physical and biological conditions that are *per se* unfavourable for land-associated human activities (Oldeman *et al.*, 1991). GLASOD includes six types of wasteland: active dunes, salt flats, rock outcrops, deserts, ice caps, and arid mountain regions (Oldeman *et al.*, 1991).

A visual representation of the relationships between the different terms is given in Figure 1.1.

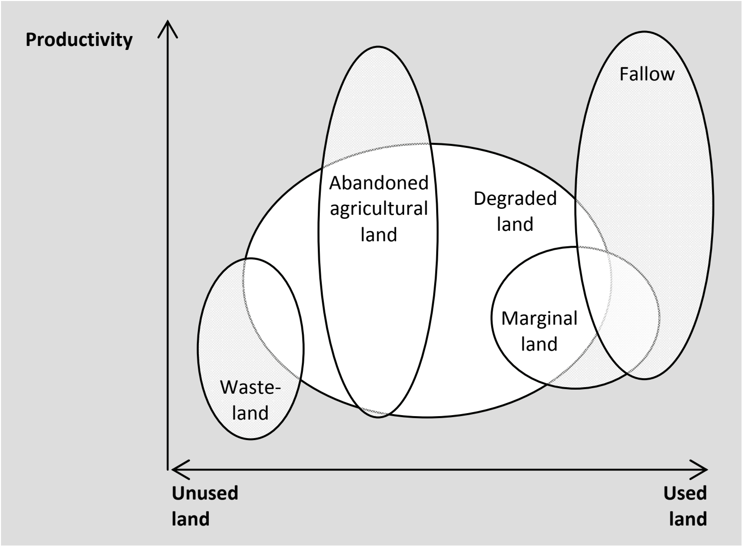

Figure 1.1: Different types of land considered in the discussion of degraded land and their relationships (adapted from Wiegmann *et al.*, 2008)

Note: The sizes of the circles do not necessarily reflect the actual extent of the land type.

Table 1.1: Literature overview of bioenergy production on degraded and marginal land at global scale

Study	**Time horizon**	**Type of land considered**	**Bioenergy production system**	**Global Extent** Mha	**Yield** t dm ha^{-1} y^{-1}	**Potential** EJ y^{-1}
Hoogwijk *et al.* (2003)	2050	Degraded land estimated from existing literature	Woody energy crops	430 - 580	1 - 10	8 - 110
Hoogwijk *et al.* (2005)	2050	Low-productive land	Woody energy crops	Not given	< 3	5 - 9
Van Vuuren *et al.* (2009)	2050	Abandoned agricultural land and natural grasslands, accounting for no to severe land degradation categories based on GLASOD	Woody energy crops	Not given [a]	Not given	104 None to minor degradation 31 Serious degradation 12 Severe degradation 147 Total
Nijsen *et al.* (submitted)	Current	Human-induced degraded land (excluding salt-affected land) based on GLASOD, excluding current forest land, cropland and pastoral land	Woody energy crops	386 [b]	Average 8.9 (2.7 – 10.1)	32
Tilman *et al.* (2006)	Current	Agriculturally abandoned and degraded land	Low intensity/ high diversity grassland cultivation	Roughly 500	4.5 [c]	45
Campbell *et al.* (2008)	Current	Abandoned agricultural land	Not defined	385 - 472	4.3 [d]	32 – 41
Schubert *et al.* (2009, based on Beringer and Lucht, 2008)	2050	Unused and degraded land	Grassy and woody energy crops	240 - 500	Not given	34 – 120

a - The potential estimates of van Vuuren *et al.* (2009) exclude land classified as forest, cropland, pastureland, bare areas, urban areas, protected areas/nature reserves.

[b] - In a background document to Nijsen *et al.* (submitted), Nijsen (2010) finds that only 21% of the degraded land area in GLASOD (excluding salt-affected soils) is not classified as forest, cropland, pastureland, bare areas, urban areas, protected areas/nature reserves. This results in 386 Mha of degraded land used in the potential analysis.

[c] - Tilman et al (2006) estimate a biomass yield of 90 GJ ha^{-1} y^{-1}. Assuming an average energy content of 20 GJ $(t\ dm)^{-1}$, this is equivalent to a yield of 4.5 t dm ha^{-1} y^{-1}.

[d] – The yield given by Cambell *et al.* (2008) refers to area-weighted mean production of above-ground biomass on abandoned agricultural land worldwide

Previous studies on the bioenergy potential of degraded and marginal land have investigated various types of degraded and marginal land (see Table 1.1) but only Nijsen *et al.* (submitted) account for the effects of different severity levels. Most studies exclude land classified as forest, cropland, bare areas, urban areas, and nature reserves from availability for bioenergy production but in the case of Hoogwijk *et al.* (2003) and Tilman *et al.* (2006) this is not further specified. The studies assessing the potential of degraded land have focused on human-induced degraded land but paid limited attention to naturally degraded land. However, these areas can be large especially when considering salt-affected soils. For example, human-induced salt-affected soils are estimated to amount to 76 Mha (Oldeman *et al.*, 1991). But naturally salt-affected soils and human-induced salt-affected soils combined are estimated between 400 Mha to 960 Mha (van Oosten and de Wilt, 2000; Wood *et al.*, 2000; FAO, 2001b; FAO, 2008b), depending on the datasets and the classification systems used.

Regarding the yield component of the potential of bioenergy production on degraded land, it is expected that yields on degraded land are lower than on other land types because of the more difficult growing conditions on degraded land. However, reclamation activities and the soil regeneration potential of trees may improve yields over time. Average woody and grassy biomass yield estimates in studies on global bioenergy potentials from degraded land and associated land types (see Table 1.1) are 1 – 10 tonne dry matter (t dm) ha^{-1} y^{-1} for degraded land (Hoogwijk *et al.*, 2003), less than 3 t dm ha^{-1} y^{-1} for low-productive land (Hoogwijk *et al.*, 2005), 2.7 – 10.1 t dm ha^{-1} y^{-1} for severely to slightly degraded land (Nijsen *et al.*, submitted), 4.5 t dm ha^{-1} y^{-1} for agriculturally abandoned and degraded land (Tilman *et al.*, 2006), 4.3 t dm ha^{-1} y^{-1} for abandoned agricultural land (Campbell *et al.*, 2008), and 2 – 5 t dm ha^{-1} y^{-1} for marginal land (Bauen *et al.*, 2009). Although it is crucial to account for the type and the severity of degradation in order to properly design energy crop production systems and to more appropriately determine the yields, only Nijsen *et al.* (submitted) do so. Other literature shows that biomass yields can in some cases of degraded and marginal land be high. For example, Metzger and Hüttermann (2008) list woody energy crops *Albizia lebbek* and *Dendrocalamus strictus* yielding 20 t ha^{-1} y^{-1} and 32 t ha^{-1} y^{-1}, respectively, on a mine spoil in a dry tropical region in India, and *Populus deltoides* yielding 4 to 20 t ha^{-1} y^{-1}, depending on the tree density, in semiarid regions close to the Thar Desert in India. McElroy and Dawson (1986) find biomass yields of 12 to 15 t dm ha^{-1} y^{-1} for short-rotation coppice willow on marginal land in Ireland. These case study results further reinforce that more systematic research on the yield of different energy crops on different types and at different severity levels of degraded land is needed to allow better estimations of bioenergy potentials and their economic performance from degraded and marginal land.

1.2.2 Economic performance of bioenergy production on degraded and marginal land

Biomass production on degraded land is likely to be more expensive than on other types of land as a result of lower yields and the reclamation activities that will likely have to be undertaken. However, biomass production costs could benefit from low land rents, so overall production costs remain unclear. Few studies have assessed biomass production costs on degraded and marginal land and its economic potential. One example is the assessment of agroforestry systems (including biomass production for fuelwood and charcoal use) on sodic soils (a type of salt-affected soils) in India (Bose and Bandoyopadhyay, 1986; Ahmed, 1991; Singh *et al.*, 1994; Singh *et al.*, 1997; Stille *et al.*, submitted). These studies find agroforestry on sodic soils to be an economically viable land use option and biomass production costs to be competitive with market prices of fuelwood and charcoal. However, these studies focus on sodic soils only, and there is little information available on other types and severity levels of degraded land. As mentioned above, the type and severity of land degradation are crucial factors in reclamation and management of degraded land and, therefore, also in production costs. In addition, it is important to account for the potential positive side effects of reclaiming degraded land (*e.g.* the regeneration of the soils or carbon sequestration) which can reduce production costs, if an economic value can be assigned. Lewandowski *et al.* (2006) showed for cadmium-contaminated land in Germany and Stille *et al.* (submitted) for sodic land in India that the economic value of the reclamation is large but both studies did not determine the effect on production costs. More research into the economic potential of bioenergy from degraded and marginal land is also required in order to determine the actual contribution it can make to (bio)energy demand.

1.2.3 Environmental impacts of bioenergy production on degraded and marginal land

Degraded and marginal land for bioenergy production is promoted because of presumed reductions in environmental impacts compared to biomass production from other land types (Section 1.2). However, bioenergy production on degraded and marginal land may also pose environmental risks to soils, biodiversity, and water. For example, degraded soils are often more susceptible to soil degradation, particularly if unsuitable species are planted or inappropriate management is applied (Wiegmann *et al.*, 2008). Furthermore, areas where degradation is related to low rainfall and/or water shortages are more susceptible to a deterioration of the situation as a result of bioenergy production (Berndes, 2002). In addition, if reclamation of degraded land is followed by an intensification of use, the conversion to bioenergy production can lead to lower levels of biodiversity, especially when degraded land has been left to natural succession for a long period of time (Schubert *et al.*, 2009). However, not enough is known about these processes to determine under what conditions (*e.g.* type and the severity of degradation, bioenergy crop, reclamation method) the use of degraded and marginal land for bioenergy production can lead to negative or positive environmental impacts. An important aspect in these considerations is also the GHG balance of bioenergy produced on degraded and marginal land, especially since this is one of the main reasons for promoting bionergy. Assessments at the case study level are particularly important for helping understand the potential environmental impacts of bioenergy production on degraded and marginal land.

In summary, the proposition of using degraded and marginal land for bioenergy production is based on presumed positive social and environmental impacts, particularly the assumption that no direct or indirect LUC will occur nor the problems associated with LUC. However, degraded and marginal land is in fact often in use, and its conversion to bioenergy production may also entail environmental and social risks. The conditions under which these risks exist and when they can be avoided are not well understood. In addition, the bioenergy production potential on degraded and marginal land on different geographical scales (from local to global) and its main components - the extent and severity of degraded and marginal land, its availability and suitability for bioenergy production, and the yields - need to be researched more thoroughly to determine the actual contribution degraded and marginal land can make to global, regional, national, and local (bio)energy demand. Finally, biomass production costs on degraded and marginal land are important in determining whether bioenergy production on degraded and marginal land is feasible, and more specific research on production costs and the economic potential of bioenergy production on different types and different severity levels of degraded and marginal land is necessary.

1.3 OBJECTIVES AND RESEARCH QUESTIONS

This thesis aims at closing some of the gaps of knowledge on bioenergy production on degraded and marginal land identified above. Therefore, the main objective is to assess the potential, the economic performance and environmental impacts of bioenergy production on degraded and marginal land in different settings and on different geographical scales ranging from the local to global scale. To this end, the following research questions are addressed:

I What is the bioenergy production potential of degraded and marginal land in different settings and on different geographical scales?

II What is the economic performance of bioenergy production and its positive side effects in different settings of degraded and marginal land?

III What are the environmental impacts of bioenergy production in different settings of degraded and marginal land?

1.4 OUTLINE OF THESIS

The research questions are addressed in Chapters 2 through 6. Each chapter evaluated bioenergy production on degraded and marginal land in different settings and geographical scales. Chapters 2 and 3 focused on palm oil production in Malaysia and Indonesia because of the recent global debate about the negative environmental impacts of palm oil. These chapters assessed the use of *Imperata* grasslands as an alternative to tropical rainforest or other land types. Chapter 4 evaluated cassava ethanol, jatropha oil, and fuelwood production from marginal semi-arid and arid land in eight sub-Saharan African countries because these regions have been under-researched and have substantially different conditions and requirements for bioenergy production than more humid regions. Chapters 5 and 6 investigated woody bioenergy production from salt-affected land focussing on a global scale (Chapter 5) and on local and national scales in Bangladesh, India, and Pakistan (Chapter 6). Salt-affected land is important to study because of its large extent and the difficulties it poses for agricultural production. Table 1.2 presents an overview of the settings, geographical scales, and the research questions that are addressed in these chapters.

Table 1.2: Overview of the settings of bioenergy production on degraded and marginal land, geographical scales, and research questions addressed in Chapters 2 through 6

Chapter	Settings	Geographical scale	Research question		
			I	II	III
2	- Palm oil production systems on *Imperata* grasslands and other land types	- Local (case study in Northern Borneo, Malaysia)			•
3	- Land use patterns and palm oil production on *Imperata* grasslands	- National (Indonesia and Malaysia)	•		•
4	- Cassava ethanol, jatropha oil, and fuelwood production on marginal semi-arid and arid land	- Sub-continental to national (eight countries in sub-Saharan Africa)	•	•	
5	- Woody biomass production from forestry plantations on salt-affected soils	- Global to sub-continental (17 world regions)	•	•	
6	- Woody bioenergy production from agroforestry and forestry plantations on salt-affected soils	- Sub-continental to local (three case studies in South Asia)		•	•

Following this introduction, ***Chapter 2*** addresses ***research question III*** by analysing the greenhouse gas balance of crude palm oil and palm fatty acid distillate production in northern Borneo (Malaysia) for different reference land use systems (including degraded land), their transport to the Netherlands and their co-firing with natural gas for electricity production. In the case of CPO, conversion to biodiesel and the associated GHG emissions are also studied. In addition, this chapter studies the effects on the GHG balance of three unresolved methodological issues: 1) how to allocate emissions to by-products, 2) the allocation period over which LUC emissions should be amortised, and 3) the choice of the fossil fuel electricity reference system. Moreover, the effects of different management options on the GHG balance are assessed.

Chapter 3 addresses ***research question III*** by compiling and analysing national level data on land use (including degraded land) and land use change and its causes in Indonesia and Malaysia over the past 30 years. This chapter also explores the role that palm oil has played in past LUC and that projected growth in palm oil production may play in LUC until 2020. Moreover, it suggests strategies to minimize negative effects. In the growth projections of palm oil production the future potential of palm oil production on

Imperata grasslands in Indonesia and Malaysia is assessed, which addresses ***research question I***.

Chapter 4 addresses ***research question I*** and ***II*** in its assessment of the current technical and economic bioenergy production potential of three bioenergy production systems (cassava ethanol, jatropha oil and fuelwood) in semi-arid and arid sub-Saharan Africa. First, the land area that is available for bioenergy production is determined, accounting for other land uses such as biodiversity conservation and agricultural production and for the suitability of land for energy crop production. Next, the crop yields and production costs of these systems are estimated and cost-supply curves are constructed. The analysis focuses on various countries with large semi-arid and arid areas. In an attempt to capture the variability in conditions found in different countries in the three geographical regions of sub-Saharan Africa the following eight countries are assessed: Kenya, Tanzania, Zambia (East Africa), Senegal, Burkina Faso, Mali (West) and Botswana and South Africa (South).

Chapter 5 addresses ***research questions I*** and ***II*** by assessing the current global technical and economic potential of woody energy crops cultivated on salt-affected land. This is done by first classifying and mapping the different types of salt-affected land and assessing its current use by applying land use/cover data. Next, a tree growth model is constructed to estimate the yields of different salt-tolerant tree species in salt-affected environments. The results of the first and second step are then combined to estimate the technical bioenergy potentials from salt affected land (***research question I***). Finally, the costs of biomass production are calculated and cost-supply curves constructed to evaluate the economic potential of energy crop production on salt-affected soils (***research question II***).

Chapter 6 addresses ***research questions II*** and ***III*** by assessing the economic and environmental performance of biosaline (agro-)forestry system with three case studies on different types of salt-affected soils in South Asia. The economic performance is assessed by studying the net present value (NPV) and the production costs; the environmental performance is assessed by studying the greenhouse gas (GHG) emissions of biosaline (agro-)forestry and exploring environmental opportunities and risks of biosaline (agro-)forestry systems in terms of biodiversity, water and soil conditions. Representing an additional source of income, the economic impact of trading carbon credits generated by biosaline (agro-)forestry is also investigated. The three case studies analyze biosaline (agro-)forestry systems in different settings: 1) a rice-tree agroforestry plantation on coastal saline soils in Bangladesh, 2) a rice-wheat-tree agroforestry plantation on waterlogged, salt-affected soils in India, and 3) a forestry plantation on saline-sodic soils in Pakistan.

Chapter 7 summarizes and evaluates the findings from Chapters 2 to 6 and provides answers to the research questions. This chapter also presents final conclusions with respect to the potential and the economic and environmental performance of bioenergy production from degraded and marginal land and gives recommendations for further research.

Chapter 2

Different palm oil production systems for energy purposes and their greenhouse gas implications

Abstract: This chapter analyses the greenhouse gas (GHG) emissions of crude palm oil (CPO) and palm fatty acid distillate (PFAD) production in northern Borneo (Malaysia), their transport to the Netherlands and their co-firing with natural gas for electricity production. In the case of CPO, conversion to biodiesel and the associated GHG emissions are also studied. This study follows the methodology suggested by the Dutch Commission on Sustainable Biomass (Cramer Commission). The results demonstrate that land use change is the most decisive factor in overall GHG emissions and that palm oil energy chains based on land that was previously natural rainforest or peatland have such large emissions that they cannot meet the 50 to 70% GHG emission reduction target set by the Cramer Commission. However, if CPO production takes place on degraded land, management of CPO production is improved, or if the by-product PFAD is used for electricity production, the emission reduction criteria can be met, and palm oil-based electricity can be considered sustainable from a GHG emission point of view. Even though the biodiesel base case on logged-over forest meets the Cramer Commission's emission reduction target for biofuels of 30%, other cases, such as oil palm plantations on degraded land and improved management, can achieve emissions reductions of more than 150%, turning oil palm plantations into carbon sinks. In order for bioenergy to be sustainably produced from palm oil and its derivatives, degraded land should be used for palm oil production and management should be improved.

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2.1 INTRODUCTION

Over the past decade, many industrialised countries have sharply increased the amount of biomass they import. This is primarily due to the fact that such countries introduced policies to stimulate renewable energy use and that imported biomass is often more cost-efficient than domestic biomass. Increasing global trade and consumption of bioenergy has been accompanied by a growing concern about the environmental, ecological and social impacts of bioenergy production. This concern has been spurred by reports about bioenergy crop production causing deforestation and the associated loss of biodiversity, greenhouse gas (GHG) emissions, displacement of forest people and related land conflicts, to name just a few. Southeast Asian palm oil, in particular, has been associated with major problems such as clear-cutting of natural rainforest, destruction of ecologically valuable peatland and instigation of social conflicts, and its sustainability has been intensely debated in many countries (Wakker, 2004; Colchester *et al.*, 2006; CREM *et al.*, 2006; Helms *et al.*, 2006). As a result of these unintended and undesired effects of bioenergy production, various initiatives have attempted to develop sustainability criteria in order to ensure sustainable bioenergy trade (van Dam *et al.*, 2006; Cramer Commission, 2007; Department for Transport, 2007; European Commission, 2007; Ryckmans *et al.*, 2007). In Europe, such efforts began in Belgium where an energy company developed its own certification system that is widely accepted by Belgian authorities (van Dam *et al.*, 2006; Ryckmans *et al.*, 2007); in the UK where, as part of the renewable transport fuel obligation (RTFO), reporting guidelines on carbon and sustainability are being developed (Department for Transport, 2007); and in the Netherlands where the so-called *Cramer Commission* on sustainable production of biomass has recently finished its work (Cramer Commission, 2007). The European Commission is also working on legislation to guarantee the sustainable production of biomass (European Commission, 2007).

In all of these initiatives, the GHG balance is an important sustainability criterion because the presumed GHG emission savings compared to fossil energy are a key driver of increasing bioenergy consumption. However, it cannot simply be assumed that bioenergy results in GHG emission savings since both the land use change (LUC) associated with biomass production and inputs needed for such LUC like fossil fuels for machinery, fertiliser and pesticides can generate GHG emissions (van Dam *et al.*, 2004; Dornburg and Faaij, 2005). LUC in particular has been found to strongly affect the GHG balance either by emissions from, for example, the net loss of standing biomass when natural rainforest is converted to other uses, or by sequestration of carbon from, for example, a net increase of soil carbon when degraded land is converted to bioenergy production (Dornburg and Faaij, 2005; Germer and Sauerborn, 2007; Fargione *et al.*, 2008; Reijnders and Huijbregts, 2008).

Although methods for calculating GHG balances have been developed for the Belgian, British and Dutch initiatives (Bauen *et al.*, 2006; Bergsma *et al.*, 2006; Ryckmans *et al.*, 2007), several aspects of implementation and verification of this sustainability criterion remain debatable. Such unsettled aspects include the method of allocating emissions to by-products, the allocation period over which LUC emissions should be amortised and the choice of the fossil electricity reference system. Moreover, these methodologies have not yet been tested on specific production cases. Therefore, the main objectives of this study are 1) to analyse the GHG balance of specific palm-oil-based energy chains and 2) to study the effects on the GHG balance of the three above-mentioned unresolved methodological issues, as well as the effects of different reference land use systems and of different management options. In order to do so, the following chains are considered:

1) *CPO electricity chain*: production of crude palm oil (CPO) in northern Borneo, Malaysia, transport to the Netherlands and co-firing at a natural gas power plant in the Netherlands;
2) *PFAD electricity chain*: production of the palm oil derivative palm fatty acid distillate (PFAD) in northern Borneo, Malaysia, transport to the Netherlands and co-firing with natural gas for electricity production in the Netherlands; and
3) *Biodiesel chain*: using the CPO for the production of biodiesel in Malaysia and transporting the biodiesel to the Netherlands for use in vehicles (Wicke *et al.*, 2007).

The GHG emission calculations are based on the methodology developed by the *Cramer Commission* since, in order for the analysed chains to be considered sustainable, they will have to meet the *Commission*'s criteria.

The remainder of this chapter is organised as follows: The methodology applied for calculating the GHG emission reductions of bioenergy compared to fossil reference systems is described (Section 2.2), and the data input is presented (Section 2.3). Then, the results of the GHG analysis of the three chains, of their various cases and of the effects of the methodological choices are presented in Section 2.4, followed by a discussion of the results and the methodological choices (Section 2.5). Section 2.6 presents the study's final conclusions.

2.2 METHODOLOGY

This study determines the greenhouse gas emissions from CPO and PFAD-based electricity and CPO-based biodiesel production according to the Dutch *Cramer Commission* methodology for GHG calculations (Bergsma *et al.*, 2006), which is based on a life cycle inventory and accounts for all GHG emissions that arise between initial land use conversion through final use of the palm oil-based energy.

The three most important greenhouse gases, carbon dioxide (CO_2), methane (CH_4) and nitrous oxide (N_2O), are included. For comparing the emissions of these three gases, the concept of global warming potential (GWP) is applied following the guidelines of IPCC, allowing for a comparison of the radiative forcing of the different gases (IPCC, 2006). The other main greenhouse gases (hydrofluorocarbons, perfluorocarbons, and sulphur hexafluoride) are not taken into account as they are insignificant in the bioenergy production chains.

The GHG emissions of by-products, which are used outside the system boundaries, are calculated on the basis of system extension. This approach assumes that the by-product generated can replace the same or a similar product that was produced from another feedstock. Due to this replacement, an emission credit for the avoided GHG emission from the original production of the product can be assigned.

The percentage of GHG emission reduction is calculated by dividing the difference in GHG emissions from the fossil and bioenergy chain by the emissions of the fossil reference system. The reduction percentage is measured against the standards set by the Cramer Commission, which requires an emission reduction of 50 to 70% for bio-electricity and 30% biodiesel in order for these to be considered sustainable (Cramer Commission, 2007). A negative percentage of emission reduction refers to a bioenergy system that has larger emissions than the fossil energy system. A positive percentage of emission reduction refers to a bioenergy system that reduces GHG emissions compared to the fossil reference system. A percentage of emission reduction of more than 100% refers to a bioenergy system that sequesters more CO_2 than is emitted in terms of CO_2 equivalent (CO_2-eq.) throughout the production chain. The functional units are defined as production of one kWh of electricity for the electricity chains and one MJ fuel for biodiesel.

In addition to the percentage of GHG emission reduction, the emissions from palm oil energy chains are also expressed in terms of carbon payback time. This is the period of time that the bioenergy feedstock needs to be grown before the LUC emissions have been offset (Dehue *et al.*, 2007). The carbon payback period is determined by dividing the net carbon loss from LUC per hectare by the amount of carbon saved per hectare and per year by the use of bioenergy (excluding LUC emissions).

For this study, case specific data from a field visit of two plantations, two mills and one refinery in the Sandakan region of northern Borneo, Malaysia are used. The field visit was conducted in connection with a Roundtable for Sustainable Palm Oil (RSPO) and *Cramer Commission* pre-audit by the certification body Control Union in February 2007. The plantations visited were well managed, demonstrated by their integrated pest management, waste minimisation and landfill practices, zero burning and habitat conservation and restoration. Each plantation had its own mill on site, but the refinery was located in Sandakan, approximately 100 km from the plantations, from where CPO and its derivative products can be directly shipped abroad. The GHG emissions of the transesterification process are based on data from the literature because the case study did not include transesterification of CPO.

2.2.1 CPO Electricity Chain

The first step in the CPO electricity chain is the land use conversion necessary to establish an oil palm plantation, followed by the production of the fresh fruit bunches (FFB), the milling and production of crude palm oil, transportation of the CPO to the Netherlands and CPO-based electricity production (Figure 2.1). Each of these steps and the resulting GHG emissions and credits are described in more detail in the following sections.

2.2.1.1 Land Use Change

Land use change (LUC) refers to the conversion of one type of land to another (*e.g.* forestland to oil palm plantation). Such a conversion affects the carbon stocks of standing biomass, belowground biomass, soil carbon and carbon stored in dead organic matter

(DOM). Various reference land use systems are studied: logged-over forest (also referred to as “base case” because it resembles the case study), natural rainforest, peatland and degraded land. The LUC emissions from aboveground biomass, DOM and soil carbon stock changes are determined for each of the land use systems based on the Tier 1 methodology of the IPCC guidelines on GHG emissions from LUC (IPCC, 2006).

The CO_2 assimilation at the oil palm plantation accounts for only the CO_2 that is fixed in the oil palm trunk and in the fronds that are not cut at harvest. This delineation is necessary so that it can be assumed that FFB and its products (CPO) and the by-products empty fruit bunches (EFB), palm kernel shells (PKS) and fibre are carbon neutral in the later steps of the production chain. Fresh fruit bunches and the fronds that are cut off at harvest are applied as organic fertiliser and dealt with in the following section.

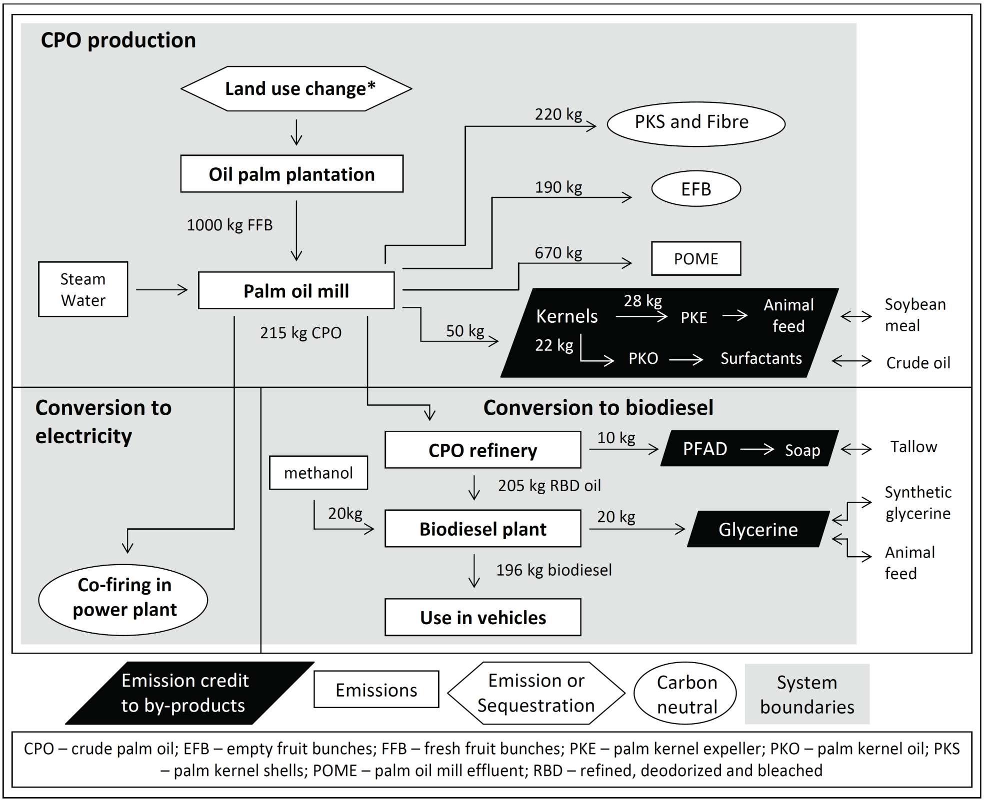

Figure 2.1: System boundaries of the two CPO-based chains with approximate mass flows and an overview of emission sources and credits. Not shown are the different transport stages that also cause GHG emissions.

* Whether CO_2 is emitted or sequestered as a result of LUC depends on the land use reference system.

For each unit of palm oil energy to account for its share of the GHG emissions from LUC and the assimilation of CO_2 by the oil palms, the net emissions from LUC are calculated by

$$\textit{LUC emissions} = 3.7 \times [(\text{LUC C} / (T_{LUC} \times Y) - C_{uptake} / (T_{plant} \times Y)] \quad (1)$$

where LUC emissions - Net emissions from LUC (g CO_2-eq. MJ^{-1}CPO); 3.7 - Molecular weight ratio of CO_2 to C (unitless); LUC C - Loss of carbon (C) from LUC (C ha^{-1}); C uptake - Carbon uptake by oil palms during plantation lifetime (C ha^{-1}); T_{LUC} - Allocation time period of LUC emissions (y); T_{plant} - Plantation lifetime (y); Y - Energy yield (MJ CPO ha^{-1} y^{-1}).

For the situation in which peatland is drained and then planted with oil palm, the additional CO_2 and N_2O emissions from peat decomposition after drainage are determined according to the IPCC guidelines for LUC (IPCC, 2006).

The displacement of prior crop production and the possible land use induced by the movement of prior crop production to other areas (indirect LUC) is not included in this study. However, this displacement may contribute significantly to the overall GHG emissions (Searchinger *et al.*, 2008).

2.2.1.2 Oil Palm Plantation

Various GHG-emitting inputs (*e.g.* diesel and fertiliser) are needed for the production of FFB at an oil palm plantation. While most of the harvest is done manually, some machinery, farm equipment and trucks for FFB transport require fossil energy and emit GHG. The GHG emissions from *fossil energy* are calculated by multiplying the amount of fuel needed per hectare of land by the emission factor of the fuel.

GHG emissions from the production of machinery and equipment, construction of buildings and production and use of pesticides are disregarded as they are minor compared to overall emissions in the system (Bergsma *et al.*, 2006).

Nitrogen (N) fertiliser applied at the oil palm plantations causes GHG emissions during its production and N_2O emissions from its application to the field. Only the GHG emissions from N fertiliser production are calculated here because the emissions from phosphate and potash fertiliser production were found to be much smaller than N fertiliser production (Bergsma *et al.*, 2006). The GHG emissions from *N fertiliser production* are calculated by multiplying the amount of a specific N fertiliser by the emission factor for producing that fertiliser.

The direct and indirect N_2O emissions from organic and inorganic *N fertiliser application* are calculated according to the IPCC guidelines for N_2O emissions from managed soils (IPCC, 2006). Since the organic fertilisers (EFB and fronds) are piled in thin layers on the ground, it can be assumed that they decompose aerobically and result in no additional GHG emissions.

The various GHG emissions from the plantation are then summed and converted to per unit of energy (MJ_{CPO}) by dividing the emissions by the FFB yield, the oil extraction rate and the energy content of CPO.

2.2.1.3 GHG Emission Flows at Mill

At the mill, GHG emissions arise from fossil fuel use (calculated as determined in the previous section) and from the palm oil mill effluent (POME), while emission credits are given to by-products. For the latter, GHG emission credits for *by-products* are only given if the by-product is used to replace another product outside the system boundaries, as is the case for kernels (Figure 2.1). *Kernels* receive GHG emission credits because they are used to produce palm kernel oil (PKO), which can then be used for surfactant production, and palm kernel expeller (PKE), which is used as animal feed and is assumed to replace soy meal. It is assumed that PKO is a feedstock in the surfactant production of alcohol ethoxylates (AE) and that, as a final product, it replaces 3-mole AE from petrochemical feedstocks. A petrochemical-surfactant-by-PKO-surfactant displacement of 1:1 is assumed based on information given by Stalmans *et al.* (1995). Credit for PKO surfactant is calculated by first determining the emission factors of crude oil surfactants and PKO surfactants. The difference in emission factors is then multiplied by the amount of surfactants that can be replaced by PKO. The second by-product, PKE, is assumed to replace soybean meal as animal feed. The GHG emission credit for PKE is calculated by multiplying the difference in emission factor of soybean meal and PKE.

At the case study site, palm oil mill effluent (POME), *i.e.* the wastewater generated from clarification and other processing steps, is treated in open ponds in order to reduce its biological oxygen demand. During the anaerobic treatment, biogas with a composition of approximately 60% CO_2 and 40% CH_4 is generated (Shirai *et al.*, 2003). The amount of carbon released as CO_2 and CH_4 is the same amount of carbon that had been sequestered during the growth of the FFB. Thus, the CO_2 from biogas is considered carbon neutral. In contrast, CH_4 from biogas has a higher GWP than the CO_2 that was initially taken up and therefore cannot be considered neutral in terms of GHG emissions. To account for the initial CO_2 uptake, the emission factor of CH_4 from POME treatment is taken to be the GWP of CH_4 (23 t CO_2-eq. t^{-1} CH_4) minus the amount of CO_2 that was taken up by the oil palm but then released as CH_4 during POME treatment, *i.e.* 2.75 t CO_2 t^{-1} CH_4. The GHG emissions from POME treatment are then calculated by multiplying this emission factor with the amount of methane produced.

2.2.1.4 GHG Emissions from CPO Transport

GHG emissions from transport encompass the *transport of CPO* by trucks to the harbour, by ocean vessel to Rotterdam, the Netherlands, and by inland ship from Rotterdam to the Claus Power Plant (Maasbracht, the Netherlands). GHG emissions from transporting CPO are calculated by multiplying the emission factor by the distance for each transportation step, adding up those emissions and then dividing by the energy content of CPO.

2.2.1.5 GHG Emissions from Co-firing CPO

The Claus Power Plant, operated by Essent uses a natural gas boiler and a conventional steam cycle, which allows co-firing of vegetable oils without major modifications to the system. Built in 1977, the Claus Power Plant has a low electrical efficiency compared to modern combined cycle natural gas power plants. The CO_2 emissions from co-firing CPO for electricity production are not accounted for in the GHG balance of CPO-based electricity as the CO_2 emitted is equal to the amount that had been taken up in producing the FFB.

2.2.1.6 Overview of CPO Production Cases

All emissions from the CPO electricity chain are converted to emissions per kWh by applying the electric efficiency of the Claus Power Plant. CPO production is studied using various land use reference systems, methodological issues such as the allocation of land use emission over different time spans and different methods for allocating emissions to products and by-products, and management improvement options for the plantation and mill (Table 2.1). In each of the land use cases (case 1 to 4), a different pre-conversion reference land use system is studied. In the management case (case 5), four management improvement options are studied in order to determine by how much the GHG emissions of the base case can be reduced. These options are:

1. Establishing new oil palm plantations on degraded land;
2. Reducing CH_4 emissions from POME: anaerobic digestion of POME takes place in a closed system so that the generated biogas can be collected more easily. In this case, CH_4 emissions from outdoor POME treatment and additional GHG emissions from replaced electricity production are avoided because the collected CH_4 can be burned for producing electricity. If the national electricity grid is close to the mill, surplus electricity could be fed into the grid, replacing electricity from other sources;
3. Increasing the oil yield by planting better tree varieties, improving harvesting techniques (*e.g.* timing and collection) and better management;
4. Applying more organic N fertiliser such as the nutrient-rich slurry from POME treatment.

In order to determine the effects of the different methodological choices, cases 6 and 7 account for different time periods over which the GHG emissions from LUC can be distributed. Cases 8, 9 and 10 analyse the effects of different methods for allocating emissions.

2.2.2 PFAD Electricity Chain

CPO refining results in refined, bleached and deodorised (RBD) oil as the main product or in its derivatives RBD stearin and olein. The only by-product of refining is palm fatty acid distillate (PFAD), which results from filtering the fatty acids and amounts to less than 5% by weight of all processed CPO. PFAD is commonly used in producing soap, animal feed, plastics and other intermediate products for the oleochemical industry (Rupilius and Ahmad, 2006). Additionally, its high energy content and the small modification that is needed to co-fire PFAD with natural gas or oil have contributed to its increasing use in power generation (Bradley, 2006). Figure 2.2 illustrates the PFAD production chain, the various sources of GHG emissions and emission allocation to the RBD oil.

Although PFAD is considered a by-product, it is an important input for the oleochemical and animal feed industries. Therefore, this analysis includes the refining process in the PFAD production chain despite a differing suggestion from the *Cramer Commission* methodology (Bergsma *et al.*, 2006). Economic allocation of the GHG emissions from the refinery to PFAD and RBD oil is applied because RBD oil is the main product and is not further used in the chain.

Table 2.1: Description of CPO production cases

Chain #	Name of case	LUC: original land type	LUC emission: allocation period (y)	Allocation / system extension	CPO/PFAD production data
Land use					
1	Base case (logged-over forest)	Logged-over rainforest	25[a]	system extension	Production data from case study
2	Natural rain-forest	Natural rainforest	""	""	""
3	Degraded	Degraded land (grassland)	""	""	""
4	Peatland	Peatland – forest cover	""	""	""
Management					
5	Management improvement	Degraded land (grassland)	""	""	CH_4 collection and electricity production, improved yields, increased organic fertiliser
Method					
6	13 year	Logged-over forest	13	""	Production data from case study
7	100 year	""	100	""	""
8	Economic	""	25	Allocation by market price	""
9	Mass	""	""	Allocation by mass	""
10	Energy	""	""	Allocation by energy	""

"" – same as above

a – While the average lifetime of a plantation is 25 years, the productive lifetime is only 21 to 23 years because no fruits are produced in the first years. The unproductive years are accounted for by averaging the FFB yield over the plantation lifetime.

Refining of CPO consumes steam and electricity, and in the case study electricity is obtained from three sources: purchased from the grid (emissions equal the amount of electricity bought multiplied by the emission factor of average Malaysian electricity production); produced onsite from biomass, *i.e.* from combustion of EFB, PKS and fibre from independent mills (the biomass streams for steam and electricity production are assumed to be carbon neutral because the emitted carbon is assumed to equal the amount sequestered by EFB, PKS and fibre during their growth); and produced onsite from fossil diesel in a generator (emissions equal the amount of fossil diesel multiplied by the emission factor of fossil diesel). Other inputs required in the refinery are bleaching earth and phosphoric acid, but both in such small quantities (7 kg bleaching earth t^{-1} CPO and 500 kg phosphoric acid kt^{-1} CPO) that the possible emissions of their production and use can be neglected.

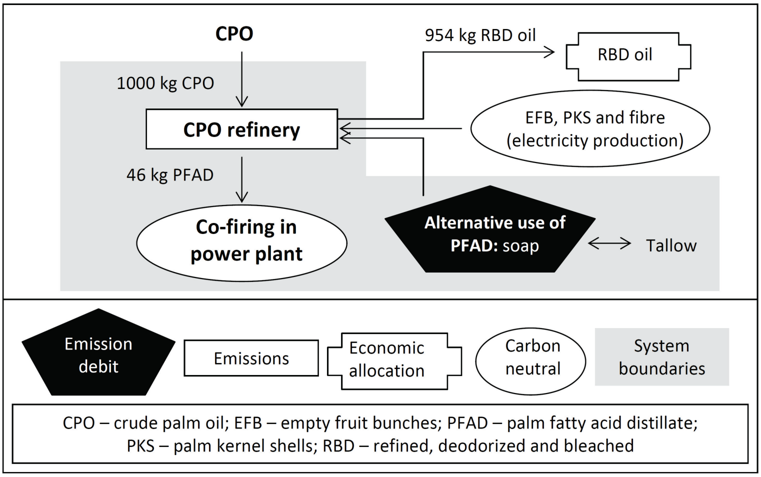

Figure 2.2: System boundaries of the PFAD electricity chain with approximate mass flows and an overview of emission sources/credits. Not shown are the different transport stages which also cause GHG emissions.

Since PFAD is currently primarily used in soap and detergent production, this analysis assumes that this is the alternative use of PFAD. It is further assumed that the PFAD for soap production is substituted by tallow from beef production as both contain mainly long chain esters and that this substitution takes place at a rate of 1:1 (by weight). Because PFAD consists of the same fatty acids as CPO, it is assumed that the 1:1 tallow to palm oil substitution ratio as applied by Postlethwaite (1995) is also valid for PFAD to tallow. The GHG emissions of the alternative PFAD use are then calculated by first multiplying the amount of tallow by the emission factor of tallow and then dividing the result by the energy content of PFAD.

PFAD is transported in the same manner as CPO, and due to similar energy content and density values of PFAD and CPO, it is assumed that the fossil energy requirements (and therefore GHG emissions) for PFAD transport to and within the Netherlands is the same as for CPO (see section 2.2.1.4). CO_2 emissions from co-firing PFAD for electricity production are not accounted for as it is assumed that CPO is produced sustainably and thus the CO_2 emitted in combustion equals the CO_2 assimilated during plant growth.

2.2.2.1 Overview of PFAD Production Cases

In addition to the PFAD base case (economic allocation, case 1) described above, three deviations are also considered. In cases 2 and 3 the emissions of the refinery are allocated on the basis of mass and energy, respectively. The case “PFAD no refinery emissions” (case 4) is based on the notion that PFAD can be treated as a residue rather than a valuable by-product. In that case only the emissions associated with PFAD treatment, transport or consumption need be accounted for. Emissions from fossil energy consumption during refining are excluded in this case.

2.2.3 Biodiesel Chain

An alternative to using CPO in electricity production is its use in the production of biodiesel. In the main process, base catalyst transesterification, the triglycerides of the oil are reacted with methanol to form methyl ester and glycerine. The biodiesel chain is composed of CPO production, CPO refining, transesterification of RBD palm oil and transport and storage at the various stages (Figure 2.1). It is assumed that CPO is first refined and the resulting RBD palm oil is used in the transesterification process because the filtering out of free fatty acids increases the oil-to-PME conversion efficiency (Meher *et al.*, 2004).

The GHG emissions of CPO production and transportation to the refinery/harbour are taken directly from the calculations described in section 2.2.1, and the emissions of CPO refining are based on the description of CPO refining in the PFAD chain in section 2.2.2. Since PFAD is not further used in the biodiesel chain, an emission credit is given. PFAD is assumed to replace tallow in soap production at a substitution ratio of 1:1. The emission credit is calculated by multiplying the emission factor of tallow with the amount of tallow that can be replaced by the production of one MJ of biodiesel.

RBD palm oil is transesterified at the refinery with the help of the catalyst sodium hydroxide (NaOH) and methanol. The GHG emissions of transesterification are from the use of fossil energy and the production and use of fossil methanol and the catalyst; an emission credit results from the by-product glycerine, which is assumed to replace synthetically produced glycerine.

The GHG emissions of biodiesel transport to the Netherlands are taken directly from the calculations described in section 2.2.1, and literature findings are used to determine the GHG emissions from biodiesel distribution within the Netherlands. While the use of biodiesel in vehicles is generally considered carbon neutral, the carbon atoms from fossil methanol still contribute to atmospheric emissions. The amount of these emissions is determined by assuming that one carbon atom in the empirical formula of PME ($C_{18}H_{35}O_2$) has its origin in fossil methanol (Bernesson *et al.*, 2003; EUCAR *et al.*, 2007).

The different CPO production systems and their effect on the GHG balance are also studied for the biodiesel chain. Here, only the variations in land use types and the management improvement cases are studied (cases 1 though 5). An additional case assumes that glycerine replaces wheat as animal feed rather than synthetically produced glycerine. This case is studied because replacing just five percent of fossil diesel with biodiesel in Europe would result in a glycerine production 30 times the size of current synthetic glycerine production in the EU (EUCAR *et al.*, 2007). Such an oversupply would cause the collapse of the glycerine market price – a development that is already being seen (Smeets *et al.*, 2005). When the price for synthetic glycerine decreases, other uses of glycerine, such as animal feed, become more economically interesting (EUCAR *et al.*, 2007). While glycerine will only be used in animal feed if it is cheaper than alternatives, these two options of glycerine uses can be seen as the upper and lower limit of emission credits given and that, when new uses of glycerine are found over time, the emission credit is likely to be within these limits (EUCAR *et al.*, 2007).

2.2.4 Fossil Reference System

In order to determine the GHG emission reductions of the different bioenergy chains, a fossil reference system is defined, its life cycle emissions determined and the emissions compared to those of the bioenergy chains. In order to study the effect of how different reference systems may affect the emission reduction and whether meeting the reduction targets is affected by the choice of reference system, several reference systems are chosen for the electricity chains: Claus Power Plant (natural gas only), average Dutch electricity, a modern natural gas power plant, a coal power plant and average EU 25 electricity. In the case of diesel, the fossil reference system is fossil diesel from European production.

2.3 INPUT DATA

2.3.1 CPO Electricity Production

Data input to *LUC* emission calculations is based on the IPCC default values for different reference land use systems (IPCC, 2006), except for the logged-over forest case where it is assumed that only 50% of the original biomass is left and that DOM carbon stock and soil carbon are similarly affected (Table 2.2). The total amount of carbon assimilated at the plantation is based on the results of field experiments in Indonesia and is 95 t C ha^{-1} (Syahrinudin, 2005).

The FFB yield at the case study plantations was 31 t FFB per hectare in 2006. This value is high compared to the national average yield, likely due to the fact that the plantations are currently at peak production. The case study yield is not applied in this study because it does not account for the first two years in which the plantation was unproductive nor for yield changes over time. Instead an average FFB yield of 25 t FFB ha^{-1} y^{-1} over the lifetime of the plantation is assumed. The oil extraction rate (OER) is 21.5% at the case study, a value also higher than the national averages, reflecting the good harvesting practices and management at the plantation. The energy content of CPO is assumed to be 36 MJ kg^{-1} (Tangsathitkulchai *et al.*, 2004).

The amount of fossil energy required at the plantation and the mill is taken from the case study and was found to be lower than data found in the literature (Damen and Faaij, 2006). The emission factors of the different fossil fuels are taken from the IPCC guidelines (IPCC, 2006).

Table 2.2: Input data for LUC

Parameter	Unit	Value	Source
Aboveground biomass (AGB) before land conversion	t dm ha^{-1}	350	IPCC (2006)
- Natural rainforest			
- Logged-over forest[a]	t dm ha^{-1}	175	Lasco (2002)
- Degraded land (*Imperata cylindrica*)	t dm ha^{-1}	6.2	IPCC (2006)
AGB at oil palm plantation after 25 years	t dm ha^{-1}	118	Syahrinudin (2005)
Carbon faction - Natural rainforest	kg C t^{-1} dm	490	IPCC (2006)
- Palm tree	kg C t^{-1} dm	400	Syahrinudin (2005)
- Grassland	kg C t^{-1} dm	400	Syahrinudin (2005)
C stocks of litter and dead wood - before conversion	t C ha^{-1}	2.1	IPCC (2006)
- after conversion	t C ha^{-1}	0	IPCC (2006)
- palm plantation	t C ha^{-1}	5.9	Syahrinudin (2005)
Soil organic C - reference (low activity clay soils)	t C ha^{-1}	60	IPCC (2006)
- Oil palm plantation[b]	t C ha^{-1}	40	Syahrinudin (2005)
Land-use system, management, input stock change factors	dimension-less	1.0	IPCC (2006)
Emission factor - C from drained peatland	t C ha^{-1} y^{-1}	10.7[c]	IPCC (2006)
- N_2O-N drained peatland	kg N_2O-N ha^{-1} y^{-1}	8	IPCC (2006)

dm – dry matter

a – Reducing AGB due to logging can range from 22 to 67% (Lasco, 2002). Here, 50% of the original biomass is assumed.

b – It is assumed that 50% of the soil carbon found in the first 100 cm is stored in the upper 30 cm.

c – In the IPCC guidelines, CO_2 emissions from peat oxidation depend on the original land type and the land type it is being converted to since different land types have different drainage depth requirements. For cropland (needing deeper drainage), a value of 20 t C ha^{-1} y^{-1} is assumed. However, if the drainage is shallower, such as for perennial tree systems, the emission factor for forest management of organic soils may be assumed, for which the IPCC gives an emission factor of 1.36 t C ha^{-1} y^{-1} (IPCC, 2006). The drainage depth of oil palm trees is commonly 60 cm (considered medium to shallow drainage) but can range from 30 cm to 2 m depending on the local conditions (Rieley and Page, 2005). In this study, the average of the two emissions (10.7 t C ha^{-1} y^{-1}) is assumed.

The amount of *N fertiliser* applied was determined at the case study plantation and is presented in Table 2.3. Although urea is drawn from several countries and ammonium sulphate from Japan, it is assumed that the emission factor from the production of both will be similar to those of European production (Ecoinvent, 2004). The direct and indirect emissions from applying N fertiliser are based on the default values given by the IPCC for the emission factor of direct N_2O emissions from managed soils, of indirect emissions from managed soils through volatisation and leaching or runoff, for the fractions of organic and synthetic N fertiliser that will volatise as NH_3 or NO_x, and for the fraction of all N fertiliser added to the soil that is lost through leaching or runoff (IPCC, 2006).

Kernels, produced at a rate of 240 kg t^{-1} CPO, are separated into 45% PKO and 53% PKE. Table 2.3 also presents the *PKO emission credit*, which is based on the average production of petroleum-based surfactants and PKO surfactants in Germany in 1996 (Patel, 1999), and the *emission credit for PKE*, which is based on average soybean production in the USA, import to and processing in the Netherlands (Damen and Faaij, 2006).

Table 2.3: Input data for CPO production

Parameter	Unit	Value	Source
EF fertilizer production			
- ammonium sulphate	kg CO_2-eq. kg^{-1} N produced	2.7	Ecoinvent (2004)
- urea	kg CO_2-eq. kg^{-1} N produced	1.3	Wood and Cowie (2004)
EF fertilizer application			
- ammonium sulphate	kg N ha^{-1} y^{-1}	70	Case study
- urea	kg N ha^{-1} y^{-1}	79	Case study
- organic fertiliser (fronds and EFB)	kg N ha^{-1} y^{-1}	31	Case study
AE PKO production	t AE PKO t^{-1} PKO	1.7	Patel (1999)
EF AE PKO[a]	t CO_2 t^{-1} AE PKO	2.7	Patel (1999)
EF AE petrochemical[a]	t CO_2 t^{-1} AE petrochemical	5.2	Patel (1999)
EF average surfactant mix[b]	t CO_2 t^{-1} surfactant mix	3.4	Patel (1999)
EF soy bean meal	kg CO_2-eq. t^{-1} soy bean oil	550	Damen and Faaij (2006)
EF PKE	kg CO_2-eq. t^{-1} PKE	155	Own calculations[c]
Energy for kernel crushing			
- electricity from grid	kWh t^{-1} kernel input	85	Tang and Teoh (1985)
- diesel for steam production	dm^3 t^{-1} kernel input	19	Tang and Teoh (1985)

EF – emission factor

a – GHG emission factors of surfactants are based on Patel (1999), who determines CO_2 emissions only, because only limited information is available on CH_4 and N_2O emissions from surfactant production.

b – In the base calculation it is assumed that one unit of PKO-based surfactant replaces one unit of petrochemical surfactant. However, it may be the case that it replaces one average-mix unit of alcohol ethoxylates (AE petrochemical, AE PKO, AE CNO). The effects of such a change will be taken into account in the sensitivity analysis of emissions from CPO production.

c – The emission factor of PKE includes the emissions from the energy input for kernel crushing that is allocated to PKE based on market prices and the emissions from transporting PKE to the Netherlands, where it substitutes soy bean meal.

The amount of methane emitted during *POME* treatment at the mill is based on the case study POME yield of 3 m^3 POME per t CPO, a biogas yield of 28 m^3 biogas per m^3 POME (Shirai *et al.*, 2003) and a 40% share of methane in the biogas (Shirai *et al.*, 2003).

Typical *transportation* types, fuels and emissions are taken from Damen and Faaij (2003), and distances are applied as found in the case study (100 km dedicated truck transport of CPO from the mill to the harbour/refinery, 17 000 km ocean vessel transport to the Netherlands and 200 km dedicated transport by inland ships to the power plant).

The *sensitivity analysis* tests those parameters of CPO production for which large ranges were found. The parameters tested and the ranges applied are shown in Table 2.4.

Table 2.4: Parameters and ranges for the sensitivity analysis of the CPO base case

Parameter	Unit	Low	Base case	High	Source Low; Source High
AGB natural rainforest	t dry matter ha^{-1}	280	350	520	IPCC (2006); IPCC (2006)
% AGB lost through logging	%	22	50	67	Lasco (2002); Lasco (2002)
Soil carbon pre-conversion	t C ha^{-1}	24	48	72	+/- 50% variation
FFB production	t FFB ha^{-1} y^{-1}	19	25	31	MPOB (2006); MPOB (2006)
EF production - ammonium sulphate	kg CO_2-eq. $(kg N)^{-1}$	0.9	2.7	7.6	Wood and Cowie (2004); Wood and Cowie (2004)
- urea	kg CO_2-eq. $(kg N)^{-1}$	0.9	1.3	4	Wood and Cowie (2004); Wood and Cowie (2004)
EF N_2O from managed soils	kg N_2O-N $(t N)^{-1}$	3	10	30	IPCC (2006); IPCC (2006)
Diesel consumption at plantation	GJ ha^{-1} y^{-1}	2.1	3.2	5.1	Schmidt (2007); Damen and Faaij (2006) citing Wambeck (2002)
Oil extraction rate	%	19	21	23	MPOB (2006); MPOB (2006)
Methane emissions from POME	m^3 CH_4 $(t CPO)^{-1}$	19.5	33.6	66.2	Chavalparit (2006); Shirai *et al.* (2003)
Emission credit - surfactant	t CO_2 $(t surfactant)^{-1}$	3.4	5.2	-	Patel (1999) ; -
Emission credit - soybean meal	kg CO_2 (t soy bean meal) $^{-1}$	275	550	825	+/- 50% variation

EF – emission factor

2.3.2 PFAD Electricity Chain

Input data for CPO *refining* are based on the data obtained during the field visit and relate primarily to the energy consumption at the refinery (Table 2.5). Economic allocation of emissions related to the refinery are based on February 2007 prices for RBD oil and PFAD as listed by the Malaysian Palm Oil Board (MPOB, 2006). Regarding the emissions of the *alternative use of PFAD*, the emission factor of tallow (107 kg CO_2-eq. t^{-1} tallow) is based on the life cycle inventory of tallow production in Switzerland conducted by Nemecek *et al.* (2004).

Table 2.5: Parameters and values for PFAD electricity chain and biodiesel chain

Parameters	Unit	Value	Source
PFAD electricity chain			
PFAD production rate	kg PFAD t^{-1} RBD palm oil	50	Case study
Energy content PFAD	MJ kg^{-1}	38.5	Erbrink (2004)
Energy requirements at Refinery			
- Diesel	MJ t^{-1} CPO	200	Case study
- Biomass	MJ t^{-1} CPO	650	Case study
- Electricity from grid	kWh t^{-1} CPO	23.4	Case study
Biodiesel chain			
PME density	kg m^{-3}	880	Vanichseni *et al.* (2002)
Conversion efficiency (CPO – PME)	kg PME t^{-1} CPO	960	Choo *et al.* (2005)
Energy requirements transesterification	kWh m^{-3} PME	250	Smeets *et al.* (2005)
Methanol emissions	kg CO_2-eq. t^{-1} methanol	786	Ecoinvent (2004)
Catalyst (NaOH)	kg CO_2-eq. kg^{-1} NaOH	1.2	Pré Consultants (2004)
Emissions from synthetic glycerine	kg CO_2-eq. kg^{-1} glycerine	9.6	Umweltbundesamt (2006)
Emissions from wheat as animal feed	kg CO_2-eq. t^{-1} feed	744	Nielsen *et al.* (2003)
Distribution of biodiesel in NL	g CO_2-eq. MJ^{-1} PME	1.1	EUCAR *et al.* (2007)
End use of biodiesel	g CO_2-eq. MJ^{-1} PME	5.9	Own calculations based on EUCAR *et al.* (2007)

2.3.3 Biodiesel Chain

Since no data on the transesterification of palm oil was collected during the field visit, the data used is based completely on the literature and described in Table 2.5. The energy requirement of CPO transesterification is based on general vegetable oil transesterification (Smeets *et al.*, 2005). The amount of methanol required (100 kg t^{-1} RBD oil) and the amount of crude glycerine produced (100 kg t^{-1} RBD oil) is based on Choo *et al.* (2005), while the amount of sodium hydroxide (6 kg t^{-1} RBD oil) required for transesterification is taken from a GHG balance analysis of rapeseed oil methyl ester (Borken *et al.*, 1999) assuming that this value also holds for PME because of the almost identical process and conversion efficiency (Smeets *et al.*, 2005). The emission factor of methanol (Ecoinvent, 2004), sodium hydroxide (PRé Consultants, 2004), synthetically produced glycerine (Umweltbundesamt, 2006) and wheat as animal feed (Nielsen *et al.*, 2003) are all based on typical production in Europe, which is assumed to be comparable to that in Malaysia. Emissions from distribution in the Netherlands is assumed to be the same as for fossil diesel (EUCAR *et al.*, 2007). Emissions from the use of biodiesel in the Netherlands are based on average emissions of biodiesel found in the Tank-to-Wheels study (EUCAR *et al.*, 2007).

2.3.4 Fossil Reference Systems

The emission factors for the different fossil reference systems are taken from other life cycle inventory studies and databases and are presented in Table 2.6.

Table 2.6: Life cycle GHG emissions of the reference fossil energy chains

Parameter	Unit	Value	Source
Claus Power Plant	g CO_2-eq. kWh^{-1}	559	Provision: own calculations based on Umweltbundesamt (2006); Use: Essent (2007)
Dutch average electricity mix (2000)	g CO_2-eq. kWh^{-1}	615	Damen and Faaij (2006)
Modern NG power plant	g CO_2-eq. kWh^{-1}	400	Umweltbundesamt (2006)
Dutch coal power plant	g CO_2-eq. kWh^{-1}	1000	Damen and Faaij (2006)
EU25 average electricity mix (2000)	g CO_2-eq. kWh^{-1}	486	Umweltbundesamt (2006)
Fossil diesel	g CO_2-eq. MJ^{-1}	88	Provision EUCAR *et al.* (2007); Use IPCC (2006)

2.4 RESULTS

2.4.1 CPO Electricity Chain

The breakdown of emissions by components shows that the most important source of GHG emissions is LUC, even when the CO_2 uptake of the oil palm plantation is accounted for (Figure 2.3). Conversion of peatland creates not only direct emissions from LUC (carbon stock changes in biomass, soil and DOM), but also emissions from the oxidation of organic peat soils, which are by themselves as large as the emissions from the entire rest of the chain. In contrast, CPO-based electricity from degraded land as well as from incorporating other management improvement options can even take up more CO_2 than emitted in the whole production chain (Figure 2.3).

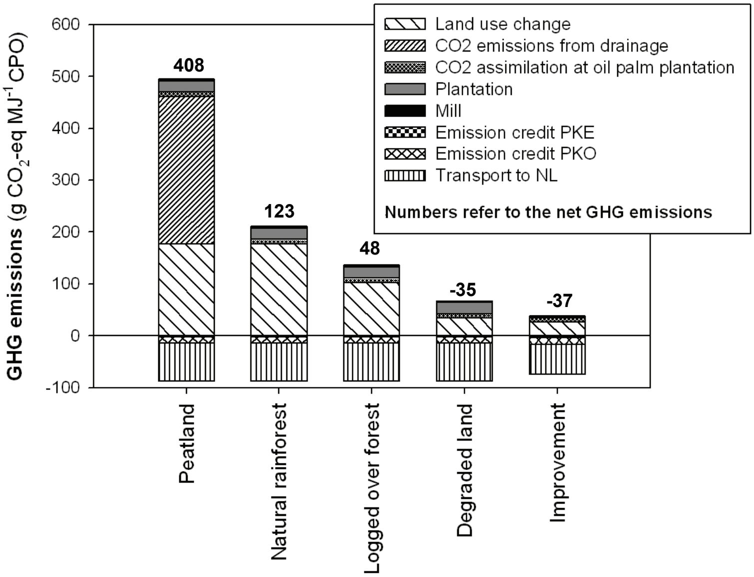

Figure 2.3: GHG emissions of CPO delivered to power plant, by source

A sensitivity analysis is conducted for individual parameters of CPO production for which the literature showed large ranges and deviations from the case study. The sensitivity analysis shows that the GHG balance is most affected by the pre-conversion aboveground biomass (AGB), percentage of AGB lost through logging and soil carbon content (Figure 2.4). The results are also, but to a lesser extent, sensitive to the amount of methane produced during POME treatment and to FFB yields. Additionally, the emission credit that is given to PKO has a large effect on the overall emissions; if the PKO-based surfactants do not replace fossil-based surfactants, as is assumed in the base case, but rather an average mix of surfactants, the overall emissions would increase by nearly 20% (Figure 2.4). In contrast, the emission credit given to PKE used as animal feed hardly affects the results. The factors that are most uncertain are the emission factors for fertiliser production, *i.e.* ammonium sulphate and urea production, and the N_2O emission factor from nitrogen fertiliser application. However, despite this uncertainty, the emission factors of ammonium sulphate and urea production scarcely affect overall emissions. In contrast, the range of the N_2O emission factor from managed soils as given by the IPCC (2006) can cause the overall GHG emissions to increase or decrease by more than 10%.

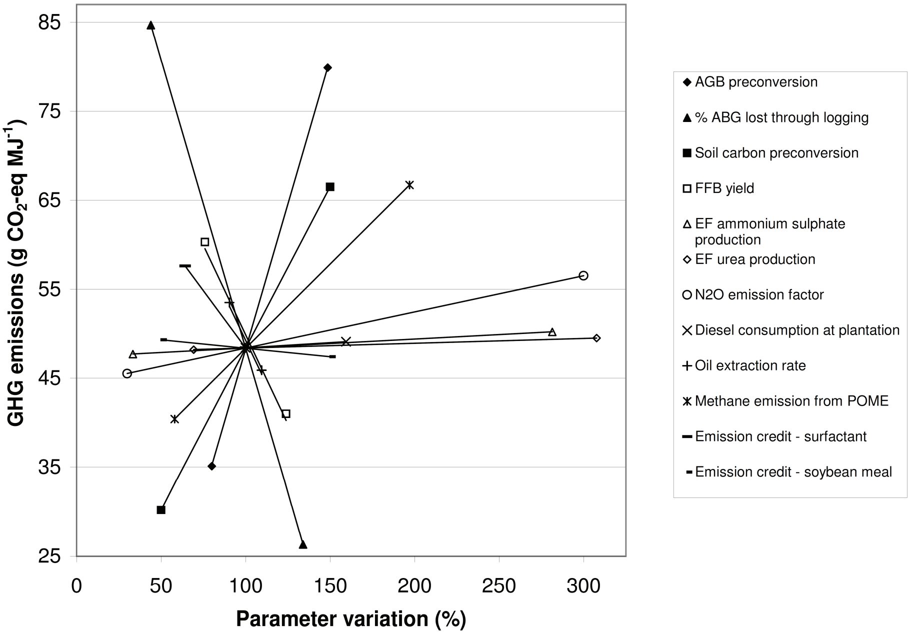

Figure 2.4: Sensitivity of GHG emissions of base case (logged-over forest)

2.4.2 PFAD Electricity Chain

The total GHG emissions of PFAD-based electricity production are only one-sixth of the emissions of the CPO base case (Table 2.7).

Table 2.7: GHG emissions of the PFAD production chain, by component

	PFAD base case	**Mass allocation**	**Energy allocation**	**No refinery emissions**
	g CO_2-eq. MJ^{-1} PFAD			
Refinery	1.6	2.2	2.2	0
Alternative use	2.8	2.8	2.8	2.8
Transport	4.2	4.2	4.2	4.2
Total	**8.6**	**9.2**	**9.2**	**7.0**

2.4.3 Biodiesel Chain

The results of the biodiesel GHG emission analysis show that the emissions of CPO used for biodiesel are in most cases lower than when CPO is used for electricity (Table 2.8). The main reason for this is the additional processing step that, using only a relatively small amount of fossil energy, produces glycerine as a by-product that, if synthetically produced, is very energy intensive and, therefore, receives a high emission credit. When glycerine replaces animal feed instead of synthetic glycerine, GHG emissions of biodiesel nearly double.

Table 2.8: GHG emissions of the biodiesel chain compared to the CPO electricity chain

	Emissions		Emission reduction		Payback time	
	Biodiesel	**CPO Electricity**	**Biodiesel**	**CPO Electricity** [a]	**Biodiesel**	**CPO Electricity** [a]
	g CO_2-eq. MJ^{-1} CPO		%		years	
Peatland forest	391	407	-337	-528	169	320
Natural rain forest	107	123	-20	-90	30	57
Base case	32	48	65	25	8	16
Base case (animal feed)	61	n/a	32	n/a	10	n/a
Degraded land	-51	-35	157	154	n/a	n/a
Improvement	-53	-37	159	156	n/a	n/a

a – compared to Dutch average electricity production

2.4.4 GHG Emission Reductions and Carbon Payback Time

The base case can meet the *Cramer Commission's* 50% emission reduction target only if it is compared to coal electricity, while palm oil electricity from degraded land and from CPO production with improved management results in emission reductions of more than 70%, regardless of the fossil reference system it is compared to (Figure 2.5). The GHG emission reductions of CPO electricity from land that was previously natural rainforest or peatland are negative, indicating that the use of CPO from these cases results in more emissions than the fossil reference systems. In contrast, PFAD-based electricity has a large potential for reducing GHG emissions (Figure 2.5).

Palm oil-based biodiesel can result in GHG emission savings above 60% if glycerine replaces synthetic glycerine and if the palm oil is not from converted natural rainforest or peatland (Table 2.8). Emissions reductions from biodiesel are significantly higher than from CPO in power production due to the emission credit given to the biodiesel by-product glycerine. If glycerine is used to displace animal feed rather than synthetic glycerine, the emission reduction drops to 32%, which is still slightly higher than the bio-electricity base case.

The carbon payback time is determined for those CPO electricity and biodiesel chains with reference land use cases in which a net carbon loss from LUC towards oil palm plantations is observed. High carbon payback periods for peatland and natural rainforest confirm that palm oil from these land types cannot be considered sustainable. In contrast, the base case on logged-over forest could contribute to GHG emission savings after eight (biodiesel) to 16 years (electricity) of palm oil production (Table 2.8).

2.4.4.1 Methodological Issues

The effects of three methodological issues on the GHG balance are presented next for the CPO electricity chain: the allocation period for LUC emissions, the method of allocating emissions to the different products and the choice of fossil electricity reference system.

These issues are expected to have a similar effect on the two other chains and are therefore not further elaborated here.

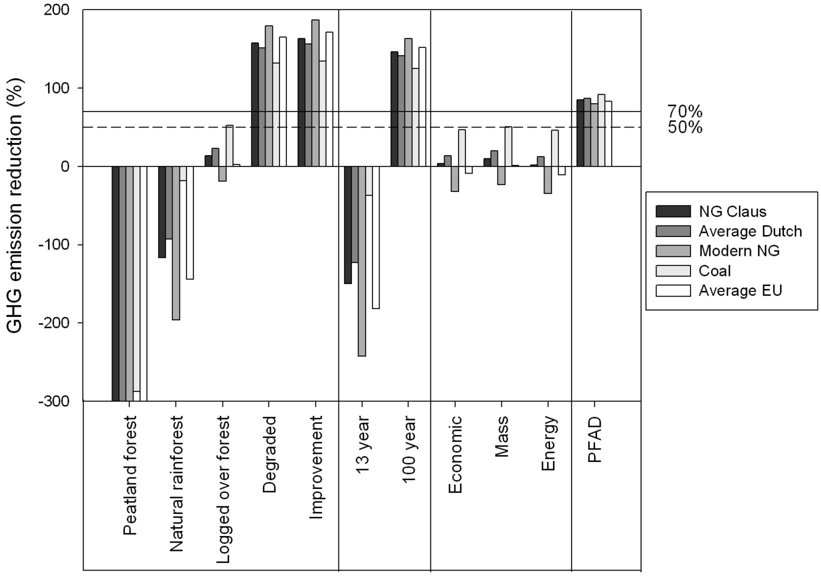

Figure 2.5: GHG emission reductions of various CPO and PFAD electricity production chains compared to different fossil reference systems

The allocation period for LUC emissions has a large impact on whether GHG emission reduction targets may be achieved (Figure 2.5). A shorter allocation period of 13 years results in negative GHG emission reductions in the base case. This was also found to be true for the other LUC cases, except when degraded land is planted with palm oil. An allocation period of 100 years results in emission reductions of more than 100% in the base case and at least 70% in other LUC cases. An exception is the peatland case, which has a negative emission reduction even with an allocation period of 100 years.

In contrast, neither the method for allocating emissions from by-products nor the choice of a fossil electricity reference system has a significant effect on the GHG emission reduction. Figure 2.5 illustrates that, although different fossil electricity reference systems do cause some variation in the bio-electricity chains’ overall emission reductions, the variation is generally not sufficient to affect whether the 50 to 70% emission reduction target is reached. Only when a case is already borderline does the fossil reference system affect whether the reduction target is met.

2.5 DISCUSSION

GHG emissions from LUC were calculated according to the Tier 1 methodology of the IPCC guidelines for national GHG inventories, which assumes that LUC does not cause a carbon stock change in belowground biomass (IPCC, 2006). However, large amounts of carbon may actually be stored in belowground biomass. While not enough data was available to have included this aspect in the main analysis of this study, it is possible to compare the carbon in belowground biomass of natural rainforest to that of grassland and oil palm plantation (based on IPCC default values for belowground biomass to aboveground biomass for natural rainforest and tropical grassland (IPCC, 2006) and on data from the field experiments of Syahrinudin (2005)). The comparison reveals that carbon in belowground biomass is 41 t C ha^{-1} for natural rainforest, 5 t C ha^{-1} for grassland and 19 t C ha^{-1} for oil palm plantation, indicating that the assumption that LUC does not cause a carbon stock change in belowground biomass is not valid. However, the inclusion of carbon in belowground biomass would not alter the general outcome of this analysis but would amplify the result that CPO production on degraded land can act as a carbon sink and that converting natural rainforest to oil palm plantations results in higher GHG emissions than a fossil-based system.

Other important aspects of the LUC issue are the displacement of prior crop production and the possible LUC induced by the movement of prior crop production to other areas or the replacement of prior crop products by alternative resources. Reinhardt *et al.* (2007) have shown that replacement of prior crop products, such as converting a coconut plantation to an oil palm plantation and substituting coconut oil with fossil oil surfactant and coconut press cake for fodder with soybean meal, causes GHG emissions that are even larger than when palm oil is produced on land that used to be natural rainforest. In such cases, *Cramer Commission* GHG emission targets could not possibly be met. Although the *Cramer Commission* has thus far excluded GHG emissions from indirect LUC from the movement of prior crop production, its sub-commission on the GHG calculation tool advises the immediate initiation of a macro-level monitoring scheme in order to investigate the effects of production displacement on the GHG balance (Bergsma *et al.*, 2006). Searchinger *et al.* (2008) recently emphasised the need for including indirect LUC in the GHG balance calculations, concluding that a focus on direct LUC would produce positive results for many chains that, when implemented, would lead to less or possibly no GHG emission reductions in reality.

The feasibility of the suggested management improvement options should also be addressed. Of the four suggested improvement options, the increased yield option is economically most interesting because of the increased income it implies. The application of more organic fertiliser is already becoming more common in Malaysia due to a new law that prohibits the direct discharge of treated POME into waterways, causing more of the nutrient-rich slurry to be spread on the plantation. However, the most effective option for reducing GHG emissions, planting oil palm on degraded land, is rare due to the fact that degraded land does not provide initial capital from timber extraction (as does natural rainforest), entails higher establishment costs and possibly reduced yields. The fourth improvement option, which is the second most beneficial option for GHG emission reduction, relates to the collection of methane from POME treatment. Currently, this option is not commonly found in the Malaysian palm oil industry, but interest in POME biogas collection and electricity production has been rising rapidly because of the possibility of getting certified emission reductions through CDM projects (Shirai *et al.*, 2003). In addition to these improvement options, other options for reducing GHG emissions should be identified and further studied.

PFAD-based electricity was found to have very low emissions compared to both fossil reference systems and to CPO-based electricity production. The most important reason for this outcome is that PFAD is treated as a by-product so only those emissions that are generated in direct connection with PFAD processing, transport and use are accounted for. Based on the mass balance of a refinery, it makes sense to treat PFAD as a by-product. However, this choice may be debatable given that PFAD is a valuable product for the oleochemical and animal feed industries. In addition, by only including emissions from the refinery process onward, PFAD-based electricity from unsustainably produced CPO could be considered sustainable. Resolving this inconsistency requires a general discussion about when to consider a product a by-product only and, in this case specifically, how to account for the possibility that unsustainable CPO may be used for PFAD production.

Based on the results of the GHG emission analysis of the electricity chains, a simple decision tree was made for determining the level of GHG emission reductions that can be reached under different conditions (Figure 2.6). This decision tree is simplified and actual compliance with GHG emission criteria depends on local conditions.

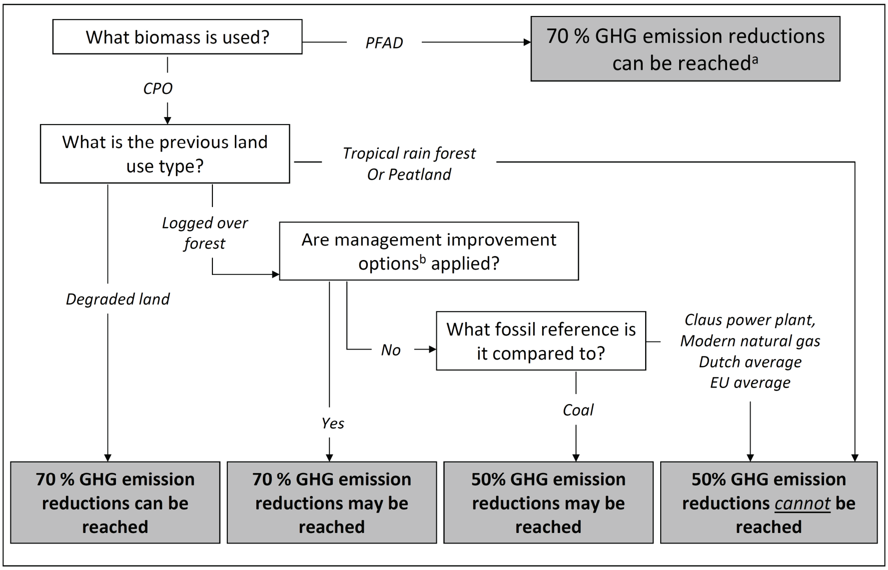

Figure 2.6: A simple decision tree for determining what emission reductions can be achieved from palm oil-based electricity production

a - Assuming that PFAD is treated as a by-product

b - The improvement options refer to 1) establishing a new plantation on degraded land, 2) increasing FFB yields, 3) treating POME in closed conditions and collecting and burning CH_4 for electricity production and 4) applying slurry from POME treatment to the plantation as organic fertiliser.

2.6 CONCLUSIONS

This study found that palm oil energy chains based on former natural rainforest or peatland have such large emissions that they cannot meet the 50 to 70% GHG emission reduction target set by the *Cramer Commission*. The case study, palm oil production on logged-over forest, can only meet the emission reduction target of 50% if compared to coal-based electricity production. However, if CPO production takes place on degraded land, the management of the production of CPO is improved (including the use of degraded land for palm oil production), or if the by-product PFAD is used for electricity production, the criteria can be met, and palm oil-based electricity can be considered sustainable from a GHG emission point of view. Even though the biodiesel base case on logged-over forest can meet the *Cramer Commission*'s emission reduction target for biofuels of 30%, other cases, *i.e.* oil palm plantations on degraded land and improved management, can achieve emissions reductions of 150% or more and can turn oil palm plantations into carbon sinks.

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Chapter 3

Exploring land use changes and the role of palm oil production in Indonesia and Malaysia

Abstract: This chapter compiles and analyses national level data on land use change (LUC) and its causes in Indonesia and Malaysia over the past 30 years. The chapter also explores the role that palm oil has played in past LUC and that projected growth in palm oil production may play in LUC until 2020 and suggests strategies to minimize negative effects. Data collection for this study revealed that the quality and quantity of data on LUC on a national scale over time are low. Despite these uncertainties, the overview of past LUC indicates that large changes in land use have occurred in Indonesia and Malaysia. In Indonesia, LUC can primarily be characterized by forest cover loss on 40 million ha (Mha) of land, a 30% reduction in forest land. Deforestation in Malaysia has been smaller in both absolute and relative terms, with a forest cover loss of nearly 5 Mha (20% reduction in forest land). Other large changes in Malaysia occurred in permanent cropland (excluding oil palm), which has decreased rapidly since the early 1990s, and in land under oil palm cultivation, which experienced a sharp increase. Projections of additional land demand for palm oil production in 2020 range from 1 to 28 Mha in Indonesia. The demand can be met to a large extent by degraded land if no further deforestation is assumed. In Malaysia, expansion projections range from 0.06 to 5 Mha, but only the lowest projection of oil palm expansion is feasible when only degraded land may be used. The role of palm oil production in future LUC depends on the size of the projected expansion as well as agricultural management factors such as implementation of best management practices, earlier replanting with higher yielding plants, and establishment of new plantations on degraded land. The current use of degraded land needs to be investigated in order to reduce possible indirect LUC, land tenure conflicts, or other social impacts. In addition to minimizing direct and indirect LUC by the palm oil sector, measures that reduce deforestation triggered by other causes must also be implemented. A key element for doing so is better planning and governance of land use, which entails more appropriate demarcation of forest land and protection of land that still has forest cover, improved monitoring of land use, and more research to uncover the complexities and dynamics of the causes and drivers of LUC.

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3.1 INTRODUCTION

The current debate about the (un-)sustainability of palm oil production in Southeast Asia has largely been spurred by land use change (LUC) that occurs by converting natural rainforest, peat swamp forest, cropland, or other land types to oil palm plantations. This LUC, in turn, has further environmental and social implications such as the loss of biodiversity, emission of greenhouse gasses (GHG) from carbon stock changes in biomass and soil, (peatland) forest fires and related respiratory diseases, and land tenure and human rights conflicts (Wakker, 2004; Colchester *et al.*, 2006; Gibbs *et al.*, 2008; Koh and Wilcove, 2008; Wicke *et al.*, 2008).

The large increase of palm oil production over the past 30 years explains in part why LUC has become a concern for the sustainability of palm oil production. The global land area of mature oil palm increased from 3.5 Mha in 1975 to 13.1 Mha in 2005. Most of this increase is found in Malaysia (increasing from 0.4 to 3.6 Mha) and in Indonesia (increasing from 0.1 to 3.9 Mha) (FAOSTAT, 2008). Including the area of immature oil palm (0.4 Mha in Malaysia (MPOB, 2008) and 1.6 Mha in Indonesia (IPOC, 2005) in 2005), a total land expansion for palm oil production of nearly 9 Mha took place in Malaysia and Indonesia between 1975 and 2005.

Case studies on a local and sometimes regional scale present detailed information on the link between oil palm expansion and LUC. For the Malaysian state of Selangor, for example, it was found that oil palm expansion was the major contributor to peatland forest fragmentation between 1966 and 1995 (Abdullah and Nakagoshi, 2007). In the state of Sabah, Malaysia, the major cause of forest disturbances shifted from logging to palm oil production (McMorrow and Talip, 2001). For the Indonesian province Riau it was determined that large scale oil palm plantations were responsible for 29% and smallholder palm oil producers for an additional 7% of the total forest cover loss between 1982 and 2007 (Uryu *et al.*, 2008). This translates into 85% of all oil palm plantations in the province being created on former natural forest land (Uryu *et al.*, 2008).

While detailed information regarding LUC as a result of palm oil production growth is available for specific locations and for some provinces, such information is sparse on a national scale. In Malaysia, expansion of palm oil production is said to have occurred primarily on logged-over forest and on former rubber and coconut plantations (Ming and Chandramohan, 2002; Abdullah and Nakagoshi, 2007), while in Indonesia natural rainforest and peatland have often been converted for palm oil production (FWI/GFW, 2002). A recent estimate by Koh and Wilcove (2008) indicates that of all oil palm expansion between 1990 and 2005 in both countries, at least 50% has come at the expense of natural rainforest. However, Koh and Wilcove do not account for other causes that triggered deforestation before oil palm plantations were established. Only a better understanding of the complexity and the dynamics of causes of forest cover loss in the past can help to prevent undesired LUC in the future.

Increasing world demand for palm oil from the food, oleochemical and energy industries combined with high prices, up to 780 $ t^{-1} crude palm oil in 2007 (MPOB, 2008), has resulted in large profits from the production of palm oil and thus an incentive for producers to expand their operations. Rising palm oil production in the future is likely to cause even more LUC and its related environmental and social impacts. In terms of the sustainability, using palm oil for energy is also discussed in combination with the GHG emissions that it causes when converting natural rainforest or peatland forest into plantations. While the GHG emissions from converting various land types to palm oil production have already been determined (Gibbs *et al.*, 2008; Wicke *et al.*, 2008), little is known about the extent to which different land types were converted to palm oil production on a national scale. The main objectives of this study are thus to 1) compile national-level data on land use over the past 30 years in Indonesia and Malaysia, 2) explore the causes of changes in land use, specifically the role that palm oil production has played, 3) investigate the extent to which future growth in palm oil production may affect LUC in both countries and 4) suggest strategies for avoiding or minimizing negative effects.

Section 3.2 of this chapter describes the methods applied for determining past LUC and its causes, explains how projections of future palm oil production growth and its land requirements were developed, and presents the input data. Section 3.3 presents an overview of past LUC in Indonesia and Malaysia, its direct causes and underlying drivers, and scenarios of possible future LUC induced by a growth in palm oil production. In section 3.4 the uncertainties of the underlying data and the assumptions that were made are discussed. Also strategies for reducing the impacts of future increases in palm oil production are described. Section 3.5 presents this chapter's conclusions.

3.2 DATA AND METHODS

3.2.1 Past land use change

An overview of LUC between 1975 and 2005 was made by collecting data of various individual land use categories from publicly available national and international statistics, government and NGO reports and academic literature. The different sources of data of the various land categories are presented and compared to each other below. The categories are *forest*, *forest plantations*, *shrubland and savannah*, *agricultural land*, *degraded land* and *other land*. An overview of the categories and data sources can be found in Table 3.1 (Indonesia) and Table 3.2 (Malaysia). The most important sources, as well as the assumptions on which the overview of past LUC is based, are described next, and an overview of the actual data used in this study is presented in Table 3.3 (Indonesia) and Table 3.4 (Malaysia).

Various sources present data on the extent of *forests* in Indonesia and Malaysia. However, large variations in the data on area covered by forests can be found. The most important reason for these differences is that some sources present data for *forest land*, while others use *forest cover*. *Forest land* refers to land that is assigned by the government to function as forest and, especially in Indonesia, can be demarcated independent of actual forest cover (Santoso, 2003). *Forest cover*, on the other hand, refers to any land that is actually covered by forest, independent of its legal demarcation as forest or non-forest. Because *forest cover* can better represent the actual status of the land than *forest land*, the category *forest* is understood here to mean forest cover. However, the concept of forest cover alone cannot properly describe the condition or quality of the forest, and this must be kept in mind when interpreting the results.

Furthermore, a distinction between the different types of forest, *e.g.* lowland forest, peatland forest, and montane forest, could shed more light into how forests are changing. However, because data availability on national scale is low, such a division was not included in the analysis. Nevertheless, since ecological impacts of converting the different types of forest to other land uses can vary, the example of Sumatra and Kalimantan is presented to provide an idea of the changes in the different types of forest (Section 3.3.1).

Table 3.1: Overview of data sources for various land use categories and their key characteristics – Indonesia

Source	Key characteristics and comments
Forest cover	
FWI/GFW (2002)	Forest cover data for 1950, 1985, and 1997. No original data but collection of data from different sources.
FAO (2006a)	Forest cover data for 1990, 2000 and 2005 based on country report (Kumarwardhani, 2005). 2005 value is based on extrapolations of trends between 1990 and 2000.
Stibig and Malingreau (2003)	Forest cover data based on satellite images of 2000 (GLC 2000); but only for one year. Includes East Timor, which is not included in the other sources.
Indonesian Ministry of Forestry (2007)	Forest cover for different governmentally assigned forest land categories for 2003 based on satellite images, but data only for one year.
Earthtrends (2007 citing GLCCD)[a]	Based on satellite images of 1992/93. Data only for one year, accuracy of only 60 to 80%.
Forest plantations	
FAO (2006b)	Forest plantation area for 1990, 2000 and 2005 based on country report.
Varmola and Del Lungo (2002)	Forest plantation area for 1986 and 1994, but no recent estimates. Extent varies strongly from FAO, 2006a.
Hooijer *et al.* (2006)	Existing and planned timber plantation concessions; no breakdown by existing vs. planned concessions, and unclear what years are referred to as "existing" and as "planned".
Indonesian Ministry of Forestry (2007)	Data presented for annual increases of forest plantations for 2000 – 2004 but no information on total area of forest plantations.
Shrubland and savannah	
Earthtrends (2007 citing GLCCD)[a] (Earthtrends, 2007)	Data for shrubland and savannah for 1992/1993 based on satellite images but data for only one year and large uncertainties (accuracy of only 60 to 80%).

Table 3.1: Continued

Agricultural land	
FAOSTAT (2008)	Agricultural land data for 1975-2005 split for total arable land, total permanent cropland and permanent pastures. No breakdown by crop; adding up area harvested of all temporary crops listed in FAO ProdSTAT (FAOSTAT, 2008) does not add up to total arable land.
Indonesian Bureau of Statistics (2007)	Agricultural land in 2005[b]. Data differs strongly from FAO data and is only available for one year.
Mature and immature oil palm	
Casson (2000)	Total area of oil palm for 1975-1999. Data is not presented for immature and mature separately; no data after 1999.
Indonesian Bureau of Statistics (2007)	Data for 1995-2006. Unclear whether data refers to area harvested or to total area; no data before 1995.
IPOC (2005)	Data for immature and mature area for 1999 to 2005; no data before 1995.
FAOSTAT (2008)	Area harvested data for 1975 – 2005; no data for immature area.
Indonesian Ministry of Agriculture (2007)	Data for area harvested 1975 – 2006. Area appears to be total area instead of harvested area because land area is significantly larger than harvested area data from FAOSTAT (2008), while 2005 data point is similar to total area by IPOC (2005).
Degraded land	
FAO (2008c; GLASOD)	Only source for estimating degraded land worldwide, breakdown by degradation type and severity level. Degraded land estimates are based on expert assessment from 1980s; overlaps possible between different types of degradation and between degraded land and other land categories.
Van Lynden and Oldeman (1997; ASSOD)	Estimates degraded land in South and Southeast Asia, breakdown by degradation type and severity level. Same approach as GLASOD (expert assessment); overlaps between different types of degradation and between degraded land and other land categories.
Casson (2000)	Data for 1998, breakdown by province. Data from Indonesian Department of Forestry and Estate Crops (Departemen Kehutana dan Perkebunan), original data and definition could not be obtained.
Indonesian Ministry of Forestry (2007)	Data for “critical land” [c] for 2002, 2004 and 2006. Unclear whether this critical land is currently used and, if so, how. Large variation between different years.
Garrity *et al.* (1997)	Area of *Imperata* grasslands. But size of overlap with other sources on degraded land or with natural grasslands is unknown; no reference year.

Note: Grey shading refers to the data that are used in the overview of LUC (Figure 3.1).

a – US Geological Survey, Global Land Cover Characteristics Database

b – Agricultural land is divided in arable land, estates and meadows with definitions similar to FAO for arable land, permanent crops and permanent pastures.

c – Critical land is defined as land which is so severely damaged that it has reduced or lost its function beyond a tolerable level (Indonesian Ministry of Forestry, 2007).

Table 3.2: Overview of data sources for various land use categories and their key characteristics – Malaysia

Source	Key characteristics and comments
Forest cover	
Economic Planning Unit (2008)	Forest area data for 1947 – 2004 for Peninsular Malaysia and Sabah, forest area data for 1988 - 2004 for Sarawak. Sarawak forest area in 1975 – 1987 is taken to be 1988 value. No definition of term "forest area", but data appears to be forest cover – 2000 value is comparable with 2000 forest cover data from (Stibig and Malingreau, 2003).
FAO (2006a; FRA 2005)	Forest cover data for 1990, 2000 and 2005 (based on Malaysia country report (Kiam, 2005)). Forest cover data is actually based on legally assigned forest land, which does not necessarily represent forest cover. The area of rubber plantation is included in forest area and in forest plantations.
Kiam (2005)	Forest cover data for 1990, 2000 and 2005. Forest data refers to legally assigned forest land and not to forest cover.
Stibig and Malingreau (2003)	Forest cover data based on satellite images of 2000 (GLC 2000); but only for one year.
Ma and Broadhead (2002)	Forest cover data for 2001. But only for one year; no reference of the information is presented.
Earthtrends (2007 citing GLCCD)[a]	Based on satellite images of 1992/93 Data for only one year and large uncertainties; (accuracy of only 60 to 80%).
Forest plantations	
FAO (2006a; FRA 2005)	Forest plantation area in 1990, 2000 and 2005. Based on country report (Kiam, 2005) it becomes apparent that FRA 2005 includes rubber plantations in both forest area (see above) and in forest plantations.
FAO (2001a; FRA 2000)	Forest plantation area in 2000. Data only for one year and different from 2000 value in FRA 2005.
Kiam (2005)	Forest plantation area in 1990, 2000 and 2005; breakdown for rubber and other timber plantations.
FAO (1984)	Forest plantation area in 1980 but only for one year.
Varmola and Del Lungo (2002)	Forest plantation area in 1990 but only for one year.
Ma and Broadhead (2002)	Forest plantation area in 2001, breakdown available for different regions. But data only for one year.

Table 3.2: Continued

Shrubland and savannah	
Earthtrends (2007 citing GLCCD)[a]	Data for shrubland and savannah for 1992/1993; based on satellite images. Data only for one year, accuracy of only 60 to 80%.
Agricultural land	
FAOSTAT (2008)	Agricultural land data split for total arable land, total permanent cropland and permanent pastures for 1975-2005. No breakdown by crop.
Mature and immature oil palm	
MPOB (MPOB, 2006; 2008)	Area of mature and immature oil palm for 1975 – 2006; breakdown for Peninsular Malaysia, Sabah and Sarawak available.
FAOSTAT (2008)	Area harvested for 1975 – 2005 but no data for immature area.
Degraded land	
FAO (2008c; GLASOD)	Only source for estimating degraded land worldwide, breakdown by degradation type and severity level. Degraded land estimates are based on expert assessment from 1980s; overlaps possible between different types of degradation and between degraded land and other land categories.
Van Lynden and Oldeman (ASSOD) (1997)	Estimates degraded land in South and Southeast Asia, breakdown by degradation type and severity level. Same approach as GLASOD (expert assessment); overlaps between different types of degradation and between degraded land and other land categories.
Sai (2002)	Data presented for marginal soils in 2002; unclear whether land is degraded or whether reference refers to marginal soil for agricultural production.
Garrity *et al.* (1997)	Data for *Imperata* grasslands. But size of overlap with other sources on degraded land or with natural grasslands is unknown; no reference year.

Note: Grey shading refers to the data that are used in overview of LUC (Figure 3.1).

a – US Geological Survey, Global Land Cover Characteristics Database

Table 3.3: Data on land use in Indonesia applied in this study; based on data from literature (*italics*) and interpolated/extrapolated results (**bold**)

Year	**Forest** (Mha)	**Forest plantation** (Mha)	**Shrubland and savannah** (Mha)	**Arable land** (Mha)	**Permanent crops w/o oil palm** [a] (Mha)	**Mature oil palm** (Mha)	**Immature oil palm** [b] (Mha)	**Permanent pastures** (Mha)	**Degraded land** (Mha)
1950	*162.3* [1]	**-**	**-**	-	-	-	-	-	**-**
1975	**130.6**	**1.4**	**10.7**	*18.0 [5]*	*7.8* [5,6]	*0.1* [7]	*0.0* [6,7]	*12.3* [5]	**3.8**
1976	**129.4**	**1.4**	**10.7**	*18.0* [5]	*7.8* [5,6]	*0.1* [7]	*0.1* [6,7]	*12.0* [5]	**5.1**
1977	**128.3**	**1.5**	**10.7**	*18.0* [5]	*7.8* [5,6]	*0.1* [7]	*0.1* [6,7]	*12.0* [5]	**6.2**
1978	**127.2**	**1.5**	**10.7**	*18.0* [5]	*7.8* [5,6]	*0.2* [7]	*0.1* [6,7]	*12.0* [5]	**7.2**
1979	**126.1**	**1.6**	**10.7**	*18.0* [5]	*7.7* [5,6]	*0.2* [7]	*0.1* [6,7]	*12.0* [5]	**8.3**
1980	**125.0**	**1.6**	**10.7**	*18.0* [5]	*7.7* [5,6]	*0.2* [7]	*0.1* [6,7]	*12.0* [5]	**9.3**
1981	**123.9**	**1.7**	**10.7**	*18.0* [5]	*7.7* [5,6]	*0.2* [7]	*0.1* [6,7]	*12.0* [5]	**9.4**
1982	**122.9**	**1.7**	**10.7**	*18.0* [5]	*7.7* [5,6]	*0.2* [7]	*0.1* [6,7]	*12.0* [5]	**9.5**
1983	**121.8**	**1.8**	**10.7**	*18.0* [5]	*7.6* [5,6]	*0.3* [7]	*0.2* [6,7]	*11.9* [5]	**9.7**
1984	**120.7**	**1.8**	**10.7**	*18.0* [5]	*7.4* [5,6]	*0.3* [7]	*0.2* [6,7]	*11.1* [5]	**9.8**
1985	*119.7* [1]	**1.9**	**10.7**	*19.5* [5]	*7.7* [5,6]	*0.3* [7]	*0.2* [6,7]	*11.9* [5]	*10.0* [9]
1986	**117.5**	**1.9**	**10.7**	*20.2* [5]	*8.3* [5,6]	*0.4* [7]	*0.2* [6,7]	*12.2* [5]	**10.1**
1987	**115.3**	**2.0**	**10.7**	*21.2* [5]	*8.7* [5,6]	*0.4* [7]	*0.3* [6,7]	*12.8* [5]	**10.3**
1988	**113.2**	**2.1**	**10.7**	*21.2* [5]	*9.4* [5,6]	*0.5* [7]	*0.3* [6,7]	*12.7* [5]	**10.4**
1989	**111.1**	**2.1**	**10.7**	*20.9* [5]	*9.3* [5,6]	*0.6* [7]	*0.4* [6,7]	*13.3* [5]	**10.6**
1990	**109.0**	*2.2* [3]	**10.7**	*20.3* [5]	*10.6* [5,6]	*0.7* [7]	*0.5* [6,7]	*13.1* [5]	**10.7**
1991	**107.0**	**2.3**	**10.7**	*18.1* [5]	*10.4* [5,6]	*0.8* [7]	*0.5* [6,7]	*11.7* [5]	**10.9**
1992	**105.0**	**2.3**	**10.7**	*18.1* [5]	*10.0* [5,6]	*0.9* [7]	*0.6* [6,7]	*11.8* [5]	**11.0**
1993	**103.1**	**2.4**	*10.7* [4]	*18.1* [5]	*10.5* [5,6]	*0.9* [7]	*0.7* [6,7]	*11.8* [5]	**11.2**
1994	**101.1**	**2.5**	**10.7**	*17.1* [5]	*11.2* [5,6]	*1.0* [7]	*0.8* [6,7]	*11.8* [5]	**11.3**
1995	**99.3**	**2.6**	**10.7**	*17.3* [5]	*11.0* [5,6]	*1.2* [7]	*0.8* [6,7]	*11.8* [5]	**11.5**
1996	**97.4**	**2.7**	**10.7**	*17.9* [5]	*10.8* [5,6]	*1.4* [7]	*0.8* [6,7]	*11.2* [5]	**11.7**

Table 3.3: Continued

1997	*95.6* [1]	**2.7**	**10.7**	*18.2* [5]	*10.5* [5,6]	*1.6* [7]	*0.9* [6,7]	*11.2* [5]	**11.8**
1998	**95.0**	**2.8**	**10.7**	*18.7* [5]	*10.3* [5,6]	*1.8* [7]	*1.0* [6,7]	*11.2* [5]	*12.0* [10]
1999	**94.3**	**2.9**	**10.7**	*19.7* [5]	*10.1* [5,6]	*1.8* [7]	*1.1* [6,7]	*11.2* [5]	**12.2**
2000	**93.7**	*3.0* [3]	**10.7**	*20.5* [5]	*8.9* [5,6]	*2.0* [7]	*2.1* [6,7]	*11.2* [5]	**12.3**
2001	**93.1**	**3.1**	**10.7**	*22.0* [5]	*8.7* [5,6]	*2.2* [7]	*2.2* [6,7]	*11.2* [5]	**12.5**
2002	**92.4**	**3.2**	**10.7**	*22.0* [5]	*8.5* [5,6]	*2.8* [7]	*1.9* [6,7]	*11.2* [5]	**12.7**
2003	*91.8* [2]	**3.2**	**10.7**	*23.0* [5]	*8.5* [5,6]	*3.0* [7]	*1.9* [6,7]	*11.2* [5]	**12.9**
2004	**91.2**	**3.3**	**10.7**	*23.0* [5]	*8.3* [5,6]	*3.3* [7]	*1.9* [6,7]	*11.2* [5]	**13.1**
2005	**90.6**	*3.4* [3]	**10.7**	*23.0* [5]	*8.1* [5,6]	*3.7* [7]	*1.8* [6,7]	*11.2* [5]	**13.3**

Sources: [1] FWI/GFW (2002); [2] Indonesian Ministry of Forestry (2007); [3] FAO (2006b); [4] Earthtrends (2007); [5] (2008b);[6] Indonesian Ministry of Agriculture (2007) [7] FAOSTAT (2008); [9] FAO (2008c); [10] Casson (2000)

a – *permanent crops without oil palm* is permanent crops (FAOSTAT, 2008) minus mature and immature oil palm area (Indonesian Ministry of Agriculture, 2007).

b – *immature oil palm* is the total oil palm area (Indonesian Ministry of Agriculture, 2007) minus mature oil palm area (FAOSTAT, 2008).

Table 3.4: Data on land use in Malaysia applied in this study; based on data from literature (*italics*) and interpolated/extrapolated results (**bold**)

Year	Forest area (Mha)	Forest plantation (Mha)	Shrubland and savannah (Mha)	Arable land (Mha)	Permanent crops w/o oil palm [a] (Mha)	Mature oil palm (Mha)	Immature oil palm (Mha)	Permanent pastures (Mha)	Degraded land (Mha)
1975	*22.9* [1]	**0.0**	**0.3**	*1.0* [4]	*3.1* [4,5]	*0.4* [5]	*0.3* [5]	*0.3* [4]	**1.0**
1976	*22.7* [1]	**0.0**	**0.3**	*1.0* [4]	*3.0* [4,5]	*0.5* [5]	*0.3* [5]	*0.3* [4]	**1.0**
1977	*22.3* [1]	**0.0**	**0.3**	*1.0* [4]	*3.0* [4,5]	*0.5* [5]	*0.3* [5]	*0.3* [4]	**1.0**
1978	*22.0* [1]	**0.1**	**0.3**	*1.0* [4]	*2.9* [4,5]	*0.6* [5]	*0.2* [5]	*0.3* [4]	**1.0**
1979	*21.6* [1]	**0.1**	**0.3**	*1.0* [4]	*2.8* [4,5]	*0.7* [5]	*0.3* [5]	*0.3* [4]	**1.0**
1980	*21.4* [1]	**0.1**	**0.3**	*1.0* [4]	*2.8* [4,5]	*0.8* [5]	*0.2* [5]	*0.3* [4]	**1.0**
1981	*20.8* [1]	**0.1**	**0.3**	*1.0* [4]	*2.7* [4,5]	*0.8* [5]	*0.3* [5]	*0.3* [4]	**1.0**
1982	*20.6* [1]	**0.1**	**0.3**	*1.1* [4]	*2.8* [4,5]	*0.9* [5]	*0.3* [5]	*0.3* [4]	**1.0**
1983	*20.4* [1]	**0.1**	**0.3**	*1.1* [4]	*2.9* [4,5]	*1.0* [5]	*0.2* [5]	*0.3* [4]	**1.0**
1984	*20.2* [1]	**0.1**	**0.3**	*1.2* [4]	*2.9* [4,5]	*1.1* [5]	*0.3* [5]	*0.3* [4]	**1.0**
1985	*20.1* [1]	**0.1**	**0.3**	*1.3* [4]	*2.9* [4,5]	*1.2* [5]	*0.3* [5]	*0.3* [4]	*1.0* [6]
1986	*20.0* [1]	**0.1**	**0.3**	*1.4* [4]	*3.0* [4,5]	*1.4* [5]	*0.2* [5]	*0.3* [4]	**1.0**
1987	*19.8* [1]	**0.1**	**0.3**	*1.4* [4]	*3.0* [4,5]	*1.4* [5]	*0.3* [5]	*0.3* [4]	**1.0**
1988	*19.7* [1]	**0.1**	**0.3**	*1.5* [4]	*3.1* [4,5]	*1.5* [5]	*0.3* [5]	*0.3* [4]	**1.0**
1989	*19.7* [1]	**0.1**	**0.3**	*1.6* [4]	*3.2* [4,5]	*1.7* [5]	*0.3* [5]	*0.3* [4]	**1.0**
1990	*19.6* [1]	*0.1* [2]	**0.3**	*1.7* [4]	*3.2* [4,5]	*1.7* [5]	*0.3* [5]	*0.3* [4]	**1.0**
1991	*19.4* [1]	**0.1**	**0.3**	*1.8* [4]	*3.3* [4,5]	*1.8* [5]	*0.3* [5]	*0.3* [4]	**1.0**
1992	*19.3* [1]	**0.1**	**0.3**	*1.9* [4]	*3.3* [4,5]	*1.9* [5]	*0.3* [5]	*0.3* [4]	**1.0**
1993	*19.2* [1]	**0.1**	*0.3* [3]	*1.9* [4]	*3.4* [4,5]	*2.0* [5]	*0.3* [5]	*0.3* [4]	**1.0**
1994	*19.0* [1]	**0.2**	**0.3**	*1.8* [4]	*3.4* [4,5]	*2.1* [5]	*0.3* [5]	*0.3* [4]	**1.0**
1995	*18.9* [1]	**0.2**	**0.3**	*1.8* [4]	*3.2* [4,5]	*2.2* [5]	*0.3* [5]	*0.3* [4]	**1.0**
1996	*18.8* [1]	**0.2**	**0.3**	*1.8* [4]	*3.1* [4,5]	*2.4* [5]	*0.3* [5]	*0.3* [4]	**1.0**
1997	*18.8* [1]	**0.2**	**0.3**	*1.8* [4]	*2.9* [4,5]	*2.5* [5]	*0.4* [5]	*0.3* [4]	**1.0**

Table 3.4: Continued

1998	*18.7* [1]	**0.2**	**0.3**	*1.8* [4]	*2.7* [4,5]	*2.6* [5]	*0.5* [5]	*0.3* [4]	**1.0**
1999	*18.7* [1]	**0.2**	**0.3**	*1.8* [4]	*2.5* [4,5]	*2.9* [5]	*0.5* [5]	*0.3*[4]	**1.0**
2000	*18.7* [1]	*0.2* [2]	**0.3**	*1.8* [4]	*2.4* [4,5]	*2.9* [5]	*0.4* [5]	*0.3* [4]	**1.0**
2001	*18.5* [1]	**0.2**	**0.3**	*1.8* [4]	*2.3* [4,5]	*3.0* [5]	*0.5* [5]	*0.3* [4]	**1.0**
2002	*18.4* [1]	**0.2**	**0.3**	*1.8* [4]	*2.1* [4,5]	*3.2* [5]	*0.5* [5]	*0.3* [4]	**1.0**
2003	*18.4* [1]	**0.2**	**0.3**	*1.8* [4]	*2.0* [4,5]	*3.3* [5]	*0.5* [5]	*0.3* [4]	**1.0**
2004	*18.3* [1]	**0.3**	**0.3**	*1.8* [4]	*1.9* [4,5]	*3.5* [5]	*0.4* [5]	*0.3* [4]	**1.0**
2005	**18.3**	*0.3* [2]	**0.3**	*1.8* [4]	*1.7* [4,5]	*3.6* [5]	*0.4* [5]	*0.3* [4]	**1.0**

Sources: [1] Economic Planning Unit (2008); [2] Kiam (2005); [3] Earthtrends (2007); [4] FAOSTAT (2008); [5] MPOB (MPOB, 2006; 2008); [6] FAO (2008c)

a – *permanent crops without oil palm* is permanent crops (FAOSTAT, 2008) minus mature and immature oil palm area (MPOB, 2006; MPOB, 2008).

For Indonesia, various sources present *forest cover* in different years (FWI/GFW, 2002; Stibig and Malingreau, 2003; FAO, 2006a; Earthtrends, 2007; Indonesian Ministry of Forestry, 2007). Data from FWI/GFW (2002) are used in the analysis because they present the most comprehensive overview of forest cover changes in Indonesia over time (1950, 1985 and 1997). The data are slightly different from, but still comparable to, FAO (2006a). The disparity can be explained by the different data sets used for extrapolation to missing years. In order to have a more recent estimate of forest cover, the results of an analysis of satellite images from 2003 by the Indonesian Ministry of Forestry (2007) are also used. Interpolations and extrapolations are then made for all other years by determining the average annual percentage change between the different years for which data are available. A number of studies also present *forest cover* data for Malaysia (Ma and Broadhead, 2002; Stibig and Malingreau, 2003; Kiam, 2005; FAO, 2006a; Earthtrends, 2007; Economic Planning Unit, 2008). The Economic Planning Unit (2008) presents data for the Malaysian *forest area* from 1947 to 2004. This source provides the largest data set on forest area and is therefore used in the overview of LUC in Malaysia. While no definition is given for forest area, it is assumed to represent *forest cover* because it does not match data of forest land by other sources (Kiam, 2005) while the forest area in 2000 is similar to the results for forest cover from remote sensing by Stibig and Malingreau (2003) for the same year. One drawback of this data set is that data are missing for Sarawak for 1947 until 1986. Since no other information could be found, the overview assumes that forest cover in Sarawak decreased at the same rate as Peninsular Malaysia and Sabah did (1.2% per year) between 1975 and 1987.

The category *forest plantations* is presented in various sources for Indonesia (Varmola and Del Lungo, 2002; FAO, 2006a; FAO, 2006b; Hooijer *et al.*, 2006; Indonesian Ministry of Forestry, 2007), with the estimates of their extent in Indonesia differing by up to several million hectares. The reasons for these variations could not be discovered from the existing data and information. FAO (2006a) data are used for Indonesia to present forest plantations in 1990, 2000 and 2005. The area in all other years is calculated by interpolating the annual average changes between the three given years. The extent of *forest plantations* in Malaysia also varies per source (FAO, 1984; FAO, 2001a; Ma and Broadhead, 2002; Varmola and Del Lungo, 2002; Kiam, 2005; FAO, 2006a). In Malaysia, forest plantations are mainly composed of natural rubber plantations as they are used for wood rather than natural rubber production. But in order to avoid overlaps with *permanent cropland* (FAOSTAT, 2008, see below), which accounts for natural rubber production, here only other forest plantations are accounted for. Data from Kiam (2005) are used to present the extent of forest plantations in 1990, 2000 and 2005 in Malaysia.

Data on *shrubland and savannah* for both Indonesia and Malaysia are presented by Earthtrends (2007 citing US Geological Survey, Global Land Cover Characteristics Database) for 1992/1993. As no other data on the land area of shrublands and savannahs could be obtained, it is assumed that their extent in both countries remained constant over time.

Agricultural land is divided into *arable land*, *permanent crops without oil palm*, *permanent pastures*, and *mature* and *immature oil palm*. Each sub-category of agricultural land is described separately below. For Indonesia, data for *arable land*, *permanent cropland* and *permanent pastures* for each year between 1975 and 2005 are given by FAOSTAT (2008). The Indonesian Bureau of Statistics (2007) also presents data for the agricultural land area in 2005, but despite comparable definitions of the different categories presented, the agricultural land area differs by 8 million hectare (Mha) from FAOSTAT (2008) data for 2005. Due to a lack of information on how data were collected, the reasons for this discrepancy could not be determined. Since data for the entire time period considered in this study are presented by FAOSTAT (2008), this source is used in the overview of LUC in Indonesia. The area of *permanent crops excl. oil palm* is determined by subtracting the area of mature and immature oil palm (as determined below) from the total permanent cropland. For Malaysia, the only available source that provides agricultural land data is FAOSTAT (2008), which is used for *arable land*, *permanent crops excl. oil palm* and *permanent pastures*.

The land area under *oil palm* cultivation in Indonesia is presented by FAOSTAT (2008), the Indonesian Ministry of Agriculture (2007), the Indonesian Palm Oil Commission (IPOC, 2005) and the Indonesian Bureau of Statistics (2007). Both IPOC (2005) and the Indonesian Bureau of Statistics (2007) present data for a short period only, which is why these studies are not used in this overview. The Indonesian Ministry of Agriculture presents data for area harvested, but analysing and comparing various data points with other sources (IPOC, 2005; Indonesian Bureau of Statistics, 2007) suggests that the data actually refer to the total area under oil palm cultivation, *i.e.* immature and mature area. Since the Indonesian Ministry of Agriculture (2007) is the only source that presents data for 1975 to 2005, it is used in this study to present the total area under oil palm cultivation. The FAOSTAT (2008) data on area harvested are used to represent mature oil palm. Subtracting the area harvested from the total area determines the area of immature palms. Comparing the results to data presented by IPOC (2005) for 1999 to 2005 reveals discrepancies of up to 45% in the immature area. This is caused by a difference in the mature area presented by FAO (2008a) and IPOC (2005), but the reasons for this discrepancy are unknown. For Malaysia, both FAOSTAT (2008) and the Malaysian Palm Oil Board (2008) (MPOB) present the area of mature oil palm for 1975 to 2005, and the data sets are identical until 1994. Thereafter the MPOB (2008) data are slightly higher than the FAOSTAT (2008) data; the reasons for this discrepancy could not be identified on the basis of the available information. Because MPOB (2008) also provides data for the area of immature oil palm for the same time period, this source is used to represent both immature and mature oil palm areas in the overview.

Determining the area of *degraded land* is complicated by the lack of a consistent definition of degraded land in the various reports, ambiguity about the year that the data refers to, and often incomplete information on the methods applied for determining the degraded land area. In addition, large overlaps with other land uses are likely as even degraded land is often used, in Indonesia for example, for subsistence agriculture and raising livestock. As a result, largely different areas of degraded land are found in literature. In Indonesia *degraded land* varies from 12 Mha (Casson, 2000; for 1998), 23 Mha (van Lynden and Oldeman, 1997– Assessment of the Status of Human-Induced Soil

Degradation in South and Southeast Asia (ASSOD); for the 1990s), 31 Mha (FAO, 2008c, Global Assessment of Human-Induced Soil Degradation (GLASOD); for the 1980s), to 74 Mha (Indonesian Ministry of Forestry, 2007; for 2004). Data from GLASOD are used in this study to determine the amount of degraded land in 1985 because it is the only available estimate for the early part of the period investigated. GLASOD, which is based on expert opinions and presents only rough estimates of the area affected, provides data about degraded land categorized into four degrees of severity of degradation (light, moderate, strong and extreme) and five levels of relative extent, *i.e.* the extent of degraded land in a given polygon (infrequent, common, frequent, very frequent and dominant) (FAO, 2008c). GLASOD's *light* and *moderate* degrees of severity and *infrequent* and *common* relative extent levels are not included in this analysis in order to reduce overlap with other land use categories. The actual extent of degraded land is determined by multiplying the degraded land area in each of the combinations of the two degrees of degradation *strong* and *extreme* and the three relative extent levels *frequent* (11-25% of the mapping unit is affected), *very frequent* (26-50%), and *dominant* (51-100%), with the average of the respective extent (18% for *frequently*, 38% for *very frequently,* and 76% for *dominant* (FAO, 2008c). Based on this approach, degraded land in 1985 is calculated to be 10 Mha. For the later part of the overview, the 1998 value is taken from Casson (2000) as approximately 12 Mha because using the estimates from the Indonesian Ministry of Forestry (2007) would result in large overlaps with other categories due to the very large area estimated as degraded land. The degraded land area in all other years is calculated with the average annual change between these two data points. In Malaysia, the GLASOD (FAO, 2008c) and ASSOD (van Lynden and Oldeman, 1997) databases provide information on the degraded land area. While GLASOD estimates a total of 5.5 Mha degraded land, ASSOD estimates a total of 25 Mha. The high estimate by ASSOD may be related to the potentially large overlaps of different types of degraded land, which are not determined in ASSOD. As a result, ASSOD data are not used in this analysis. The actual extent of degraded land according to GLASOD is calculated as described for Indonesia and amounts to 1 Mha in Malaysia. Because no other information could be found, it is assumed that the degraded land area in Malaysia remains constant over time. These simplifications indicate that only crude estimates of degraded land and its changes over time can be made for both countries.

Imperata cylindrica, also known as alang-alang in Indonesia and lalang in Malaysia, is often associated with degraded land in tropical Asia (Garrity *et al.*, 1997; Syahrinudin, 2005). It is the most common weed in the tropics, where it invades land that was previously inappropriately managed and then abandoned (Syahrinudin, 2005). It is unclear whether the degraded land data presented above accounts for this type of grasslands. Because of the possible overlaps and the lack of information on the extent of such overlaps, (*Imperata*) grassland is not presented as an individual category but is rather covered partially by the *degraded land* category and by the *other land* category. To provide a complete picture, a brief overview of the extent of *Imperata* grassland is given next. The most comprehensive study with overviews of *Imperata* grassland area and distribution in tropical Asia (Garrity *et al.*, 1997) indicates that 8.6 Mha of sheet *Imperata* grasslands exist in Indonesia and approximately 0.2 Mha in Malaysia. Garrity *et al.* (1997) suggest that the land area covered by *Imperata* grassland is increasing, but little is known about the rate and location of the increase.

Peatland is not considered as an individual land category in this overview because of large overlaps with other categories, particularly forest and agricultural land, and a lack of data to determine the extent of these overlaps. Even though it is not included in the overview, it is important to mention changes in peatland areas and the use of peatland given the large negative environmental impacts, especially GHG emissions, of converting peatland forest to other uses. This is done in the discussion in section 3.4.1.

The category *other land* refers to all land that does not belong to the categories described above and includes urban and built-up land, part of the previously mentioned grassland, land affected by fires and deforested land lying idle. The latter three land types are included in the *other land* category as far as they are not already accounted for in the *degraded land* category. The land area in the *other land* category is determined by taking the difference between the total land area of Indonesia (or Malaysia) and the area of all other land categories combined.

3.2.2 Causes and Drivers of LUC

The factors influencing general LUC in Indonesia and Malaysia can be divided into direct causes and underlying drivers. A qualitative overview of the causes and drivers of LUC is made by examining government and NGO reports and academic literature.

3.2.3 Projections of future land use change

The projections of future LUC are made for 2020 and are based on two components, the future palm oil production expansion and the reference land use. Future palm oil production is projected on the basis of four different data sets for each country in order to determine a range of possible future land expansion by the palm oil industry. All projections apply a constant annual percentage increase to determine oil palm expansion by 2020:

1) The *past trends* projection for both Indonesia and Malaysia is an extrapolation of past trends in oil palm expansion. The past trends are based on the period 1997 to 2005 and show an average land expansion of 10.0% per year for Indonesia (Indonesian Ministry of Agriculture, 2007) and 4.3% per year for Malaysia (MPOB, 2006).
2) The *FAO* projection for both Indonesia and Malaysia is based on the FAO forecast that palm oil production volume will increase by 5.9% per year until 2010 in Indonesia and by 3.8% per year in Malaysia (FAO, 2003b). This projection assumes that the production volume continues to increase by the same percentages until 2020.
3) The *IPOB/MPOB* projection is based on future expansion as estimated by the national palm oil associations. The Indonesian Palm Oil Board (IPOB, 2007) projects an increase of the area occupied by palm oil production of 1.5 Mha by 2010. The projection applied in the present study assumes that the land area continues to

increase at the same rate (4.6% per year) until 2020. The Malaysian Palm Oil Board (MPOB) projects the total area of palm oil production in Malaysia in 2020 to be 5.1 Mha (Jalani *et al.*, 2002), which is equivalent to an increase of 3.1% per year.

4) The *provincial plans* projection for Indonesia is based on provincial plans for future oil palm expansion (Colchester *et al.*, 2006), which estimate the total land under oil palm plantations in 2020 to be nearly 20 Mha. This is equivalent to an increase of 10.7% per year. The fourth projection for Malaysia is based on the *9th Malaysia Plan*, the Malaysian governmental plans for economic development between 2006 - 2010 (Economic Planning Unit, 2006), and projects a 5.5% increase in palm oil production per year until 2010. For this projection, it is assumed that annual production continues to increase at the same rate until 2020.

For each of the four projections, two cases are studied. In the first case, the *base case*, projected land expansion in 2020 is either taken directly from the reference of the projection (projections 1, 3 and 4 (Indonesia only)), or calculated from the projected production volume presented in the reference (projections 2 and 4 (Malaysia only)) by applying a crude palm oil (CPO) yield that is extrapolated from yield trends in the past. The palm oil yield in Indonesia fluctuated annually, but the five year averages between 1980 and 2005 were similar, which indicates stagnating yields (FAOSTAT, 2008). Thus, the 2020 yield in the *base case* amounts to 3.5 t CPO ha^{-1} y^{-1}. Between 1980 and 2005, the Malaysian yields increased by 0.7% per year (MPOB, 2006), which results in a yield of 4.3 t CPO ha^{-1} y^{-1} in the *base case* in Malaysia in 2020. In addition to yields, the *base case* also refers to a similar share of immature palms (on an area basis) as in the past, *i.e.* 29% in Indonesia and 10% in Malaysia. The second case, here after referred to as the *improved case*, assumes that the same amount of CPO is produced as in the *base case* but at improved yields and therefore on less land. It is assumed that yields can be improved by 3% each year (Dros, 2003) so that yields in 2020 will amount to 5.9 t CPO ha^{-1} y^{-1} in Indonesia and 6.1 t CPO ha^{-1} y^{-1} in Malaysia. The share of immature palms is assumed to make up 20% of the total oil palm area in both countries, as is suggested to be appropriate for continuous renewal of a plantation (Jalani *et al.*, 2002). Given that the annual yield increase of 3% is significantly larger than past yield trends, the discussion section will further elaborate on whether such increases are achievable by 2020. The land requirements and the expected production volumes of each projection and for both cases are presented in Table 3.5.

The second component of future land use change is the reference land use, which defines the land types that are allowed to be converted to oil palm plantations and determines how much land from each land category may become available for conversion to oil palm plantations in the future. The reference land use is defined for four land categories: 1) forest covered land and 2) agricultural land (excluding oil palm), as these are the two main land categories; 3) degraded land, as this land category is seen as an important factor in determining whether sustainable expansion is possible; and 4) forest plantations, as this land category is likely to be an important competitor for land. Due to limited data availability and the large overlap with other land categories, peatland forest is not included in the analysis of future LUC.

Table 3.5: Additional land requirements for palm oil production and production volume in 2020 under four expansion projections for Indonesia and Malaysia

	Total production 2020 (million t CPO)	**Additional land requirements 2005 – 2020** [a] (Mha)			
Reference land use approach	*Independent of approach and case*	*Business as Usual*		*Sustainability*	
Case		Base	Improved	Base	Improved
Indonesia					
Past trends	57	17.5	6.2	25.0	8.9
FAO	31	7.0	0.9	10.0	1.3
IPOB	31	5.2	0.9	7.4	1.3
Provincial plans	63	19.8	7.4	28.3	10.6
Malaysia [b]					
Past trends	27	3.9	2.4	3.7	2.3
FAO	26	3.0	1.5	2.8	1.6
MPOB	17	1.1	0.0	1.2	0.1
9th Malaysia Plan	33	4.9	3.2	4.7	3.1

a – Original land use by the palm oil sector in 2005 is 5.5 Mha in Indonesia (IPOC, 2005) and 4.1 Mha in Malaysia (MPOB, 2008).

b – Land requirements in Malaysia's *Business as Usual* reference land use are larger than in *Sustainability* because more degraded land is used in the former. As yields are assumed to be lower on degraded land than on other land types, reduced production is compensated for by applying more land.

Two extreme systems of reference land use are defined in order to reflect possible ranges in future developments in both countries:

1) The *Business as usual* approach assumes that land use change continues as in the past; *i.e.* forest cover loss, agricultural land changes and forest plantation growth are extrapolated according to past trends. Palm oil production, as well as other agricultural production and forest plantations, may use land from any land use category.
2) The *Sustainability* approach assumes that deforestation is stopped, agriculture increases at the same rate as projected to be likely in East Asia until 2030 by FAO (2003b) and forest plantations increase as in the past (because no other information is available). New oil palm plantings as well as any other expansion by agriculture and forest plantations are required to be located on degraded land.

Both approaches apply constant annual percentages to project changes in future reference land use. The land made available from the different land categories based on these reference land use systems is presented in Table 3.6.

Table 3.6: Land that may become available for agriculture (including palm oil production) and forest plantations between 2005 and 2020

	Indonesia		**Malaysia**	
Land made available from	*Business as Usual* (Mha)	*Sustainability* (Mha)	*Business as Usual* (Mha)	*Sustainability* (Mha)
- Forest covered land	38.7	0	0.5	0
- Agricultural land [a]	-6.4	-2.2	0.6	-0.1
- Forest plantations [a]	-2.4	-2.4	-0.2	-0.2
- Degraded land [b]	16.4	13.3	0.9	0.9
Total available land	46.3	8.7	1.8	0.6

a **–** The negative values in the *agricultural land* and *forest plantations* categories refer to additional land requirements by these categories in the future. These land requirements have to be satisfied before meeting the land demand of palm oil production and are, therefore, subtracted from available land.

b – Degraded land in Indonesia in 2020 differs in the two reference land use systems because in the *Business as usual* system degraded land increases at the same rate as in the past, while for the *Sustainability* system it is assumed to remain constant. The area of degraded land in Malaysia is assumed not to change because of the lack of information on changes in the past.

The *Sustainability* approach applies only degraded land for expansion because this can relieve the pressure on natural rainforest, and palm oil production on degraded land can function as a carbon sink (Wicke *et al.*, 2008; Chapter 2 of this thesis). The use of degraded land for palm oil production may result in lower yields, but the actual reduction depends strongly on the type of the degraded land and the severity of the degradation (Corley and Tinker, 2003). An example is the above-mentioned *Imperata* grasslands, which are found on degraded sites as well as on soils with moderate to high fertility (Garrity *et al.*, 1997). Thus, if the grass is successfully removed, good yields without additional agrochemical inputs are possible on moderately and highly fertile land invaded by *Imperata cylindrica*. In contrast, in the case that soils are degraded, special treatment with fertilizers and other agrochemical input may be required (Corley and Tinker, 2003). However, this may in turn cause additional environmental and economic impacts that must be accounted for and which may not necessarily result in yields as high as from non-degraded land. No specific information on palm oil yields on degraded land could be found in the literature, but in order to show the effects of possible yield reduction on degraded land this study applies two options: 1) a yield reduction of 30% compared to yields on other land types and 2) no yield reduction.

The available land as determined in the reference land use systems (Table 3.6) is matched with the land requirements of each projection for palm oil production growth (Table 3.5) in order to determine what these projections mean in terms of LUC and its impacts. The matching is made on a national scale and, in both reference land use systems, land demand by agriculture and forest plantations is met prior to meeting land demand by palm oil production.

3.3 RESULTS

3.3.1 Overview of Past LUC

The overview of LUC over the past 30 years in Indonesia and Malaysia indicates large changes in land use in both countries (Figure 3.1). The largest change in Indonesia (Figure 3.1, left) has occurred in forest covered land, which decreased from 130 Mha in 1975 to 91 Mha in 2005, while agricultural land increased from 38 Mha in 1975 to 48 Mha in 2005. Approximately half of this agricultural expansion is due to an expansion in palm oil production, namely from 0.1 Mha in 1975 (0.6 Mha in 1985) to 5.5 Mha in 2005, and even further to an estimated 7 Mha in 2008 (Indonesian Ministry of Agriculture, 2007). The other half of the expansion was caused by an increase in arable land, mostly for expansion of paddy rice. On a national level, the large increase in palm oil production is small compared to the 39 Mha of forest cover loss since 1975 (or 29 Mha if considering deforestation only since oil palm expansion has really started in the mid-1980s) and indicates that there are other important causes of forest cover loss and LUC in general.

However, three aspects merit consideration. First, regional trends may be different because palm oil production has been focused primarily on Sumatra and Kalimantan, where 95% of the land used for palm oil production is located (Indonesian Ministry of Agriculture, 2007). Thus, while on a national scale palm oil production does not appear to be the most important cause of deforestation, for certain regions and sub-regions it is likely to play a major role. For example, forest cover loss on Sumatra and Kalimantan (2.5% per year between 1985 and 1997) is significantly higher than Indonesia’s national-level forest cover loss (1.9% per year). More locally, Uryu *et al.* (2008) find that 65% of the Sumatran province of Riau’s forest cover was lost between 1982 and 2007, which is equivalent to 4.1% forest cover loss per year, although it should be noted that the figures are not directly comparable to the national and regional levels because of a different time frame. Second, oil palm is established primarily in lowland forest areas, which this study does not distinguish from other forest types because no national-level information is available. Instead, changes in different forest types in Sumatra and Kalimantan are briefly described here. The 2.5% forest cover loss per year in Kalimantan and Sumatra between 1985 and 1997, although higher than the average national forest cover loss, is still much lower than lowland forest loss in the same period (7% per year; own calculations based on (FWI/GFW, 2002)). Third, forest cover alone cannot account for the quality of the standing forest as large areas of forest are likely to have been used for logging. Of the 90 Mha of forest cover in 2005, only about 49 Mha are considered primary forest, of which the absolute and relative extent have been decreasing (FAO, 2006a).

In Indonesia, the changing area of the *other land* category cannot be explained on the basis of the available data because it is unclear what type of land actually experiences these changes. But it can be speculated that the increase is due to more deforested land lying idle, while recent decreases may be due to increasing use of already deforested land for agricultural expansion.

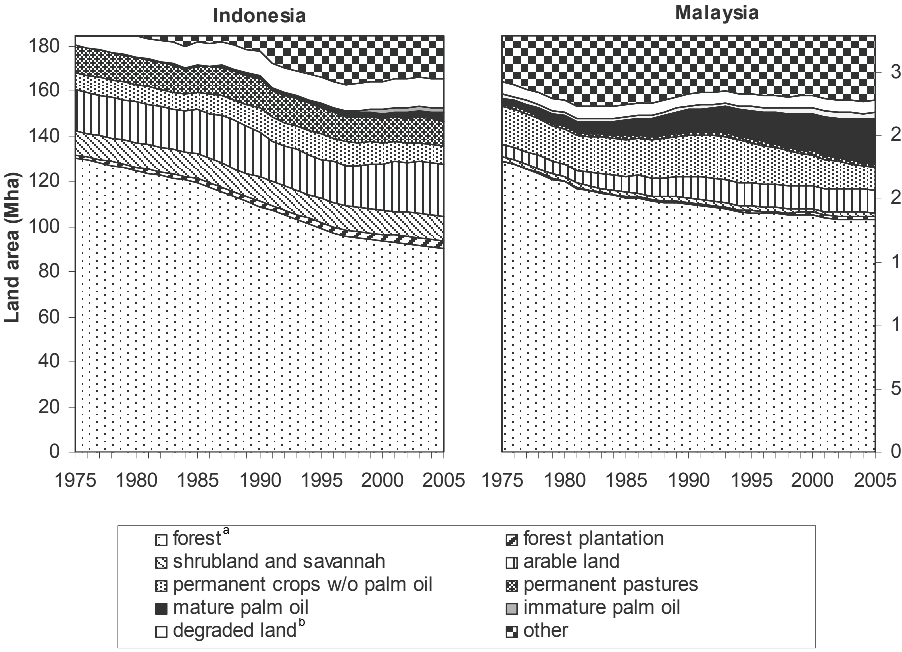

Figure 3.1: LUC in Indonesia (left) and Malaysia (right) between 1975 and 2005.

a – When interpreting Figure 3.1, it must be kept in mind that forest cover alone cannot properly describe the condition of the forest because large differences can exist among forests, especially with respect to canopy cover and tree height. While these differences can be natural, they are often due to logging or other exploitation by humans. Of the 90 Mha of Indonesian forest cover in 2005, only about 49 Mha are considered primary forest, of which the absolute and relative extent have been decreasing (FAO, 2006a). For Malaysia, primary forest cover amounted to 3.8 Mha in 2005 and has remained constant since 1990 (FAO, 2006a).

b - Due to the extrapolations and overlap of some categories with others (mainly degraded land), the sum of all land categories in Indonesia in the years 1975 to 1979 is larger than the total land area of Indonesia. As the extent of degraded land is most uncertain, the excess land is subtracted from the degraded land.

In Malaysia, LUC has also been large, but different from that in Indonesia (Figure 3.1, right). While deforestation was rapid until the beginning of the 1980s, it has, on average, slowed down since then. However, the annual rate of forest cover loss has fluctuated. A rate of 1% or more was seen in the years 1994 and 2001, while rates as low as 0.01% were observed in some other years (Figure 3.1, right). Although forest cover is still greater than 50%, of the 18 Mha of forest covered land in 2005, it is estimated that only 3.8 Mha is primary forest (FAO, 2006a). The largest change in land use was seen in oil palm cultivation increasing from 0.6 Mha in 1975 to 4 Mha in 2005. At the same time, the area of other permanent crops, primarily the export crops natural rubber and coconut, decreased significantly (FAOSTAT, 2008). On a national scale, the increasing land use by oil palm cultivation cannot be directly linked to decreasing land use by other permanent crops because available data are not spatially and temporally explicit enough. Nevertheless, case studies and anecdotal information confirm that oil palm expansion in Malaysia has often replaced other permanent crops (McMorrow and Talip, 2001; Ming and Chandramohan, 2002; Abdullah and Nakagoshi, 2007). In addition, logged-over forest is often mentioned as a land type that is converted to oil palm plantations in Malaysia. Logged-over forest first transitions from the *forest* to the *other land* category – when crown cover and the height of the leftover trees become too low to be considered a forest – and then to the *oil palm* category. Thus, the use of logged-over forest for palm oil production may explain why the *other land* category decreases over time. However, the data are not sufficient to verify this for Indonesia.

3.3.2 Causes and drivers

As shown in Figure 3.1, palm oil production alone cannot explain the large loss in forest cover in Indonesia. Instead, a web of interrelated direct causes and underlying drivers appears responsible. Literature finds important direct causes of LUC to be logging, oil palm expansion and other agricultural production and forest fires (Dauvergne, 1993; Sunderlin and Resosudarmo, 1996; FWI/GFW, 2002; Hooijer *et al.*, 2006). Also the underlying drivers of LUC in Indonesia are diverse. The two primary drivers are 1) agriculture and forestry prices, which generate more income via (il)legal logging and via palm oil production compared to other agricultural crops (Chomitz *et al.*, 2007), and 2) policy and institutional factors such as financing foreign debts by exploiting natural resources, privatization of timber and tree crop estates, corruption, and land tenure conflicts (Sunderlin and Resosudarmo, 1996; Kartodihardjo and Supriono, 2000). Domestic population growth, along with the governmentally organized and spontaneous transmigration to the outer islands (Whitten, 1987), and economic growth were also drivers of LUC in Indonesia.

The causes of forest cover loss in Malaysia vary per region. In Sabah and Sarawak, the most important causes have been timber extraction and shifting cultivation, while in Peninsular Malaysia, and in recent years increasingly in Sabah, forest cover has been affected most by conversion to agriculture and more specifically to oil palm plantations (McMorrow and Talip, 2001). The main reason for such distinctive causes in the different Malaysian regions is a result of the autonomy of Malaysian states in terms of land use and resource policies (McMorrow and Talip, 2001). The underlying drivers in Malaysia are similar to those in Indonesia in that agricultural and forestry prices, economic growth and policy and institutional factors have played a role in LUC (Drummond and Taylor, 1997; McMorrow and Talip, 2001), although domestic population growth has not been a driver of LUC in Malaysia (McMorrow and Talip, 2001). Policy and institutional factors that have affected Malaysian LUC include the orientation of policy toward using natural resources (timber and tree crops) to finance foreign debts, corruption in the allocation of timber and tree crop concessions, and land tenure conflicts (Drummond and Taylor, 1997; McMorrow and Talip, 2001; Colchester *et al.*, 2007).

3.3.3 Future projections of land use change

The projections of future land use change in relation to palm oil production expansion (Figure 3.2) show that 1) in the *Business as usual* approach much larger expansion of palm oil production is possible than in the *Sustainability* approach but that this comes at the expense of forest cover; and 2) that many projections of palm oil production expansion, especially in the *Sustainability* approach, are not feasible because they require more land than can be made available (white stacks in Figure 3.2).

The matching of projected land requirements for palm oil production with land made available in the *Business as usual* approach demonstrates that all projections for Indonesia, even those with large expansion, are possible (Figure 3.2 upper left quadrant). However, this comes at the expense of forest cover. Oil palm expansion in Malaysia is projected to be much smaller than in Indonesia, but at the same time, less land is available for expansion there. In the *Business as usual* approach feasible projections are the Malaysian Palm Oil Board (MPOB) *base case* and *improved case* and the FAO *improved case* (Figure 3.2 lower right quadrant).

In the *Sustainability* approach, less land is available than in the *Business as usual* approach because only degraded land is considered available for the expansion of agricultural production and forest plantations. This results in fewer projections that are feasible in both countries. Whether a projection is feasible in the *Sustainability* reference land use system also depends on the yield that can be achieved on degraded land. In Indonesia for example, if there is no yield reduction on degraded land, all but the two largest projections (past trends and provincial plans, both *base case*) are feasible. However, if yields are reduced, only two projections are feasible (FAO and IPOC, both *improved case*) (Figure 3.2 upper left quadrant). In Malaysia, only one projection is feasible (MPOB *improved case*), regardless of whether yields are the same or lower than on former forest or agricultural land (Figure 3.2 lower left quadrant). Even though only the lowest expansion projections are feasible in the *Sustainability* approach in both countries, the 2020 production volume in the lowest projection for Indonesia (IPOB projection) is still double the production of 2005 and 30% larger for Malaysia (MPOB projection).

While in Indonesia the total land expansion for palm oil production may be reduced by up to 50% if yields are increased, this reduction amounts to only 20% in Malaysia. The main reason for this smaller reduction is that Malaysia currently has a low share of immature palms (10%) and that, in order to keep improving yields, it is necessary to increase this share by replanting earlier. As a result, the area of immature oil palms in the *improved case* increases compared to the *base case*. In contrast, Indonesia currently has a 29% share of immature palms, which is assumed to decrease in the improved case to a share of 20%.

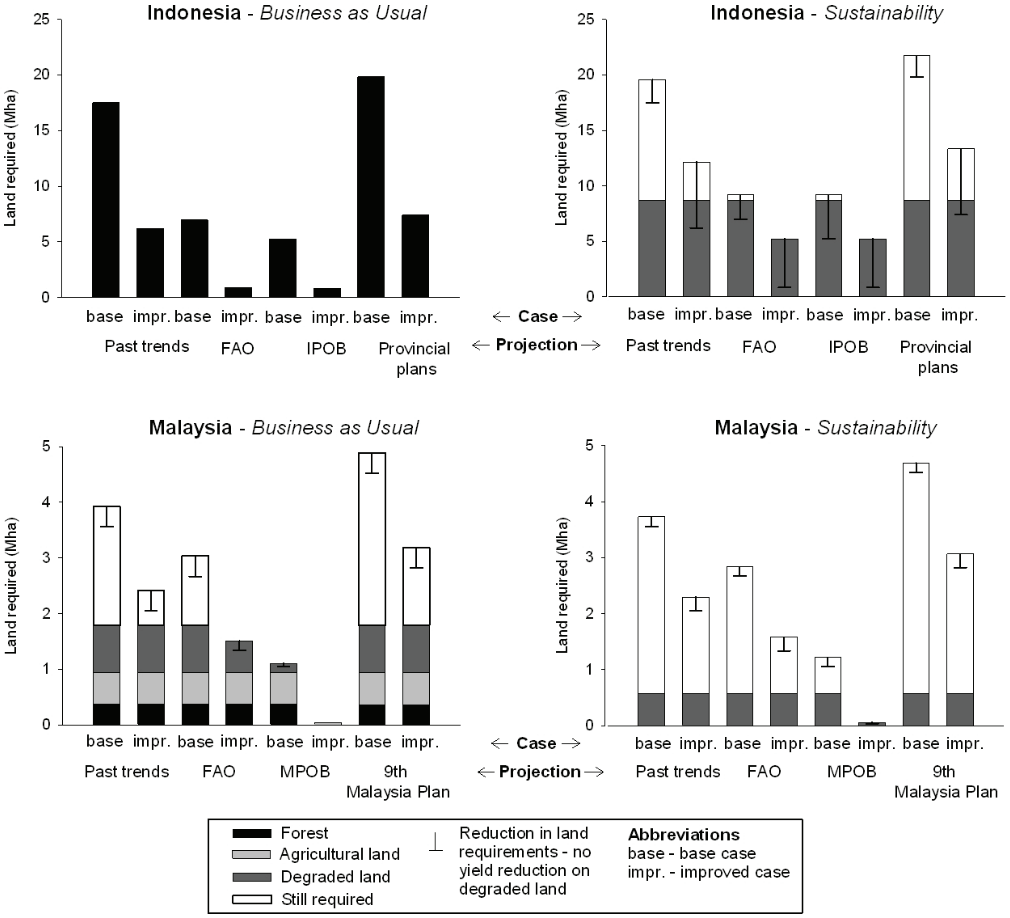

Figure 3.2: Matching land requirements (including land with immature palms)[a] with land availability in the Business as Usual reference land use system and in the Sustainability reference land use system for Indonesia and Malaysia

a - The total height of each bar refers to the total additional land requirements (assuming 30% yield reduction when using degraded land compared to other land types), while the black, light grey and dark grey stacks together refer to the available land. The white stacks refer to the land area "still required" for palm oil production expansion and indicate those cases that are not feasible with respect to land requirements. The error bar indicates the reduction in land requirements if no yield reduction on degraded land is assumed.

3.4 DISCUSSION

3.4.1 Data availability and quality

Data collection for this study revealed that much data on land use/cover and information on their changes on national scale over time is lacking and that there are large differences in the quality of the data. The categories *forest*, *degraded land*, and *other land* are most affected by uncertainties in the data. Various *forest* data sets for both countries are available, but the data sources are not always in agreement. Reasons for differences can be that legally assigned forest land is presented instead of actual forest cover, that the quality of satellite images differs (including the uncertainty caused by high cloud cover), or that different reference years are used in extrapolations. Other times the differences could not be explained because the methods for generating data were not clear. Information on *degraded land* was based on two data points for Indonesia (from the GLASOD database for 1985 (FAO, 2008c) and Casson (2000) for 1998) and only one data point for Malaysia (from the GLASOD database for 1985), all of which are only crude estimates of the extent of degraded land and must be treated as such. This problem becomes apparent when adding up the land areas of the different land categories, which results in a larger total area for the period 1975-1979 than actually exists in Indonesia. This is due to the overlap of some categories with others and due to interpolations and extrapolations when only few data points are available. Since the extent of degraded land and its overlap with other categories are most uncertain, the excess land is subtracted from degraded land. Also uncertain is the extent of the category *other land* in both countries because it depends on the extent of all other categories (as *other land* is determined by taking the difference between the total land area and the area of all other categories combined). In addition to the uncertain extent of the *other land* category, it is also difficult to determine which land types are actually changing in this category.

Data unavailability, the lack of clear definitions, and the difficulties in directly linking different causes and effects inhibited establishing the exact share of palm oil production expansion or, in this respect, of any other single cause of forest cover loss or other LUC on a national scale in Malaysia and Indonesia. The problem of establishing exact shares of responsibility for a change in land use is exacerbated by the inter-linkages that exist between causes. An illustrative example is the Indonesian phenomenon of logging and clear-cutting natural rainforest on palm oil concessions although oil palms are then never planted. This phenomenon has been described in many references (Casson, 2000; Kartodihardjo and Supriono, 2000; FWI/GFW, 2002; Colchester *et al.*, 2006), and the extent of the problem appears to be large but is not actually known. When determining the shares of responsibility in LUC, the question arises whether this deforestation is allocated to the palm oil sector or to the logging companies. Despite such allocation problems, other studies have attempted to assign shares of responsibility to the various causes of deforestation on a national scale for Indonesia. An overview of such studies made by Sunderlin and Resosudarmo (1996) shows the share of responsibility in deforestation by estate crops, primarily large oil palm plantations, to vary between 2% and 28%. This large range underlines the above-mentioned complexity in determining causes.

The analysis of past LUC and the projections of future palm oil production expansion did not include peatlands because of limited information about its past, current, and future use and because of the difficulties in establishing the extent of overlap with other land types. Although it is not included here, accounting for peatland use in future palm oil production expansion is important from a sustainability point of view. This is particularly the case for GHG emissions because the use of peatland in palm oil production causes significantly larger emissions than the already high emissions of converting natural rainforest (Wicke *et al.*, 2008; Chapter 2 of this thesis). Therefore, existing information is briefly described here. Peatland forest cover has decreased from 18 Mha in 1985 to 14 Mha in 2000 in Indonesia and from 1.6 Mha in 1985 to 1.1 Mha in 2000 in Malaysia, while in both countries peatland use for timber and oil palm plantations has increased (Hooijer *et al.*, 2006). Hooijer *et al.* (2006) attempted to determine the use of peatland by palm oil producers and found that of the “existing and planned” 10.3 Mha of palm oil concessions in Indonesia, 27% (approximately 3 Mha) is located on peatland. While this figure could have been used for Indonesia, the “existing and planned” palm oil concessions data presented by Hooijer *et al.* (2006) have several drawbacks. It is unclear what years are referred to by “now” and “planned”, the data cannot be split up for these two points in time, and the source of these plans is not defined. In addition, information regarding planned use of peatlands in Malaysia is not available.

3.4.2 Yield improvements

Palm oil yield improvements can greatly reduce land requirements, but it is debatable whether average annual yield improvements of 3% - as is assumed in the *improved case* of the projections - are actually achievable. The reasons for doubt are the stagnant yields during the past 30 years in Indonesia and an annual yield increase of less than 1% in the same time period in Malaysia (FAOSTAT, 2008). However, considering that the projected improved yields for Indonesia and Malaysia are only slightly higher than a good commercial yield of 5.5 t ha^{-1} y^{-1} already obtained on some plantations in Malaysia (Jalani *et al.*, 2002) but significantly lower than best yields obtained from breeding trials of 10 t ha^{-1} y^{-1} (Corley and Tinker, 2003) and the theoretical yield of 18 t ha^{-1} y^{-1} (Corley and Tinker, 2003), improving the current average yield seems possible in both countries. This also holds for yields on degraded land, where, with appropriate management, yield reduction compared to non-degraded land may be avoided. However, increased agrochemical use could, in turn, negatively affect the environment, particularly through nitrous oxide emissions from fertilizer application. Another aspect that must be considered is the additional costs created by the increased management and agrochemical use, which may result in lower profitability of degraded land.

If the suggested yield improvements are to be realised in both countries, strategies with which this can be done need to be determined and implemented. The most important strategies at existing plantations are to follow best management practices, including applying fertiliser and other agrochemical inputs more precisely; to practice good harvesting standards; and to transport the fruit quickly to the mill (Jalani *et al.*, 2002). Earlier replanting with higher yielding palms is also effective. This is especially an issue in Malaysia, where the share of immature palms has decreased from 40% in 1975 to 10% in 2000 (MPOB, 2006) and the share of old palms (> 25 years) has increased from less than 1% in 1975 to nearly 8% in 2000 (Jalani *et al.*, 2002). The result is lower production from older trees and slower penetration of the new, higher yielding planting materials

(Jalani *et al.*, 2002). Planting higher yielding palms is also the most important strategy for new plantations to achieve high oil yields. While these strategies can help improve yields, reaching the yield target for 2020 as proposed in the projections will largely depend on how quickly and widespread these strategies are implemented.

3.4.3 Degraded land

Because of potentially positive environmental impacts, reduced competition for land and a significantly improved GHG balance, the *Sustainability* reference land use applies only degraded land for conversion to oil palm plantations. As a result, the feasibility of each oil palm expansion projection largely depends on the availability of degraded land in addition to the actual extent of the production expansion (in terms of volume) and the yields. However, the amount and the availability of degraded land and the severity of degradation are uncertain, making it difficult to determine the feasibility of a projection. Moreover, it should be kept in mind that even if large amounts of degraded land exist, an expansion of palm oil production on such a scale may not be considered sustainable for other reasons such as the establishment of monocultures or the possible displacement of current uses such as grazing or subsistence farming. If degraded land is already in use, its conversion to oil palm plantations would cause the displacement of existing activities. Thus, in order to avoid land use and land ownership conflicts with the current users and indirect LUC by forcing these users to move into other, possibly forested areas, an assessment of land ownership and current use of degraded land needs to be made. Such an assessment can then provide information on the actual availability of degraded land for palm oil production.

3.4.4 Oil palm expansion beyond 2020

This study made projections of oil palm expansion only until 2020 because longer term projections become increasingly uncertain. However, global vegetable oil (and particularly palm oil) demand is expected to continue to increase beyond 2020. For example, Corley (2003) projects global palm oil demand in 2050 to range between 93 and 256 million t CPO y^{-1} (compared to 34 million t CPO y^{-1} in 2005, (MPOB, 2006)). As a result, it is likely that even more land will be required for palm oil production in Indonesia and Malaysia after 2020. This study found that, depending on the projection and the reference land use, in several Indonesian cases enough land is available to meet the demand until 2020 without further forest losses or replacement of other agricultural production. However, this does not hold for almost all Malaysian cases nor will this hold for most Indonesian cases if oil palm expansion is considered beyond 2020. Nevertheless, increasing yields to 5.9 t CPO ha^{-1} y^{-1} in Indonesia and 6.1 t CPO ha^{-1} y^{-1} in Malaysia, as assumed in this study, and converting only degraded land (8.7 Mha in Indonesia and 0.6 Mha in Malaysia) to oil palm, would already cause an additional palm oil production of 51.1 million t CPO y^{-1} in Indonesia and 3.5 million t CPO y^{-1} in Malaysia compared to 2005. However, the sustainability (with respect to criteria other than land availability) of converting such large amounts of land to oil palm must be evaluated, although it is beyond the scope of this article to do so.

3.5 CONCLUSIONS

This study compiled and analyzed national level data on land use change (LUC) and explored its causes, particularly the expansion of palm oil production, in Indonesia and Malaysia over the past 30 years. Data collection for this study revealed that much data on land use/cover and information on their changes on national scale over time is lacking and that there are large differences in the quality of the data. Despite these uncertainties, the overview of past LUC indicated that large changes in land use have occurred in Indonesia and Malaysia. In Indonesia, LUC can primarily be characterized by forest cover loss on 40 Mha of land (30% reduction of forest land). Land use for palm oil production increased from 0.1 Mha in 1975 to 5.5 Mha in 2005. In Malaysia, deforestation has been less severe in both absolute and relative terms, with a forest cover loss of 4.6 Mha (20% reduction of forest land). Other large changes in Malaysian land use occurred in permanent cropland (excluding oil palm), which has decreased rapidly since the early 1990s. In contrast, land use for oil palm cultivation experienced a sharp increase from 0.7 Mha in 1975 to 4.0 Mha in 2005 in Malaysia.

Although this study could not quantify the exact role that palm oil production has played in past LUC in Indonesia and Malaysia due to uncertainties in past LUC data, the literature review indicates that palm oil production has played a significant role in LUC in some areas and that this role varies among different scales and regions. Moreover, other factors, especially logging, also play an important role in national-level deforestation. A better understanding of the dynamics (*i.e.* the chronological chain of land uses) and the complexity of the various causes and drivers (*i.e.* inter-linkages between oil palm expansion and other causes and drivers) of LUC on a regional and national scale is essential for developing a more problem-specific and effective land use policy. In order to do so, the availability and quality of land use data need to be improved. This is possible through more remote sensing activities as well as more regionally specific analysis including data collection and verification at sites in question.

This study also explored the role that projected growth in palm oil production may play in future LUC. The results indicate that additional demand for palm oil in the future (until 2020) can, in many scenarios, be met without further forest cover loss by a combination of converting degraded land and improving yields. More specifically, the projections of total production in 2020 in Indonesia range from 31 to 63 million t CPO per year. In the most optimal situation, which includes converting only degraded land and improving yields from 3.4 t CPO ha^{-1} y^{-1} in 2005 to 5.9 t CPO ha^{-1} y^{-1}, oil palm expansion would be limited to 1 Mha. However, if the key condition of yield improvements is not met, land expansion for palm oil production can increase to 28 Mha. This demand can be met with deforested land if LUC continues as in the past (*Business as Usual* approach) or to a large extent by degraded land, which amounts to 12.5 Mha, if no further deforestation is assumed (*Sustainability* approach). In Malaysia, projections of total production in 2020 range from 17 to 33 million t CPO per year. In an optimal situation (*i.e.* yield increases to 6.1 t CPO ha^{-1} y^{-1} and earlier replanting) this would require no additional land, but in the worst case it would require up to 5 Mha of additional land for palm oil production. In most projections in the *Business as Usual* approach, this demand can be met by a mix of deforested land, agricultural land and degraded land. In the *Sustainability* approach, however, only the lowest projection of oil palm expansion is feasible because degraded land amounts to just 1 Mha.

In both countries, the role of palm oil production in future LUC depends primarily on the size of the projected expansion, the achieved yield and the kind of land converted (*i.e.* whether new plantations are being established on degraded land only). But achieving the yield increases suggested in this study depends heavily on whether best management practices for palm oil production are implemented, whether earlier replanting with higher yielding planting materials takes place, and the type and quality of land that is converted to oil palm plantations. The use of degraded land for palm oil production should be combined with an investigation of current uses and ownership of degraded land in order to avoid indirect LUC, land tenure conflicts and other possible environmental and social impacts. Further research on expected palm oil yields on degraded land, how they can be improved and what the impact could be on social and environmental conditions is also necessary.

Based on the insights of this study as well as the results of the analysis of GHG emissions of palm oil production from different land types and management systems (Wicke *et al.*, 2008; Chapter 2 of this thesis), a climate and forest-friendly palm oil production expansion up to 2020 is possible in principle. However, palm oil demand is expected to continue growing after 2020, and it will become increasingly difficult to sustainably meet this demand. In addition, the right incentives must be given for the expansion to take place in a sustainable manner. Enhancing the sustainability of palm oil production expansion may be achieved by incorporating the above-mentioned strategies for improving the impact of palm oil production growth on LUC as well as its GHG emissions, most prominently the use of degraded land and better management, in sustainability certification systems such as RSPO. In addition, measures that reduce LUC, especially deforestation, and degradation of land resulting from other direct causes and underlying drivers also need to be implemented. A key element for doing so is better planning and governance of land use, which entails, among other things, more appropriate demarcation of forest land and protection of land that still has forest cover, improved monitoring of land use, and more research to uncover the complexities and dynamics of the causes and drivers of LUC. Another measure would be to include the REDD (reduced emissions from deforestation and degradation) mechanism in the post-2012 climate change regime.

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Chapter 4

The current bioenergy production potential of semi-arid and arid regions in sub-Saharan Africa

Abstract: This chapter assesses the current technical and economic potential of three bioenergy production systems (cassava ethanol, jatropha oil and fuelwood) in semi-arid and arid regions of eight sub-Saharan African countries. The results indicate that the availability of land for energy production ranges from 2% (1.3 Mha) of the total semi-arid and arid area in South Africa to 21% (12 Mha) in Botswana. Land availability for bioenergy production is restricted mainly by agricultural land use, but also by steep slopes and biodiversity protection. The current total technical potential for the semi-arid and arid regions of the eight countries is calculated to be approximately 300 PJ y^{-1} for cassava ethanol production, 600 PJ y^{-1} for jatropha biodiesel or 4,000 PJ y^{-1} for fuelwood. The analysis of economic potentials shows that in many semi-arid regions, cassava ethanol, jatropha oil and fuelwood can compete economically with the reference energy sources. However, fuelwood, jatropha oil, and cassava ethanol production costs in most arid regions of sub-Saharan Africa are often above average national market prices of gasoline, diesel, and fuelwood. Nevertheless, for example, in arid Kenya 270 PJ could be produced annually with fuelwood at production costs of less than 3 US$ GJ^{-1}. Despite high production costs, it is important to investigate and invest in sustainable bioenergy production in semi-arid and arid regions of sub-Saharan Africa because of its potential to drive rural economic and social development.

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4.1 INTRODUCTION

Bioenergy is the primary source of energy for approximately 2.7 billion people around the globe, and it plays a vital role in meeting local energy demand in many developing countries (OECD/IEA, 2010b). In particular, sub-Saharan Africa is heavily reliant on biomass energy; 81% (some 653 million people) of the region's population rely on traditional biomass fuels for cooking and heating (OECD/IEA, 2010b). This figure is projected to increase to 720 million in the year 2030 (IEA, 2006). In some sub-Saharan African countries, biomass accounts for 70 to 90% of primary energy supply and 95% of total energy consumption (Karekezi, 2002). The bulk of bioenergy consumed in sub-Saharan Africa is traditional biomass such as fuelwood, charcoal, agricultural residues and animal waste (Karekezi, 2002). The traditional use of biomass in combination with often-inefficient stoves has many disadvantages including the significant amount of time spent, mainly by women and children, on fuelwood collection; indoor air pollution and deforestation and soil degradation, which is a particular problem for charcoal production in areas surrounding major cities (Karekezi, 2002; OECD/IEA, 2010b). In addition, the dependence on traditional biomass and the lack of modern energy carriers appears to be linked directly to poverty; as income levels decrease, more traditional biomass is consumed by a larger amount of the population (Karekezi, 2002; Amigun *et al.*, 2008; OECD/IEA, 2010b). As a result, solutions to the problems of traditional biomass must also account for the underlying problem of poverty.

Sustainable bioenergy production – whether in the form of modern energy carriers such as transport fuels or electricity, or in traditional energy forms such as fuelwood and their more efficient use – can reduce energy poverty, contribute to rural development and avoid the negative impacts discussed above. In addition, bioenergy production can generate employment and additional income and, thereby, reduce poverty (Bekunda *et al.*, 2009). Other important benefits of sustainable bioenergy production include the diversification of agricultural markets, increasing local production of energy and reducing dependence on costly, imported fuels while also decreasing GHG emissions (Bekunda *et al.*, 2009).

Alongside these benefits, there are also risks associated with bioenergy production in sub-Saharan Africa. Most important is the potential competition for land and other resources between food and fuel production. Although food insecurity is a subset of the larger issues of insecurity in the household and national economy, and while it is true that food security can be high even in countries with low domestic food production, the issue is a particular concern in sub-Saharan Africa, which is the most undernourished region in the world (FAO, 2008d). In addition, the corruption and weak governance often found in many sub-Saharan African countries can exacerbate the risk of food-fuel competition and may also lead to other negative effects such as deforestation and land use conflicts. Political instability in some countries in the region is also a risk factor for investment in bioenergy production.

Despite these risks, various bioenergy initiatives are already underway in sub-Saharan Africa. Examples include the jatropha electrification project in Mali, bioethanol production from sugarcane in Malawi, sisal waste used for biogas production in Tanzania and cassava-based ethanol production in Benin (Smeets *et al.*, 2009b; Smeets *et al.*, 2009a; Watson, 2009b). Furthermore, national policies for bioenergy have been or are being developed in, for example, Botswana, Burkina Faso, Cameroon, Gambia, Ghana, Kenya, Liberia, Sierra Leone, South Africa, Tanzania and Zambia (COMPETE Project, 2009).

Several studies have already shown that sub-Saharan Africa exhibits high (technical) potentials for bioenergy production (Marrison and Larson, 1996; Hoogwijk *et al.*, 2005; Smeets *et al.*, 2007). However, these studies have not paid specific attention to semi-arid and arid regions, which account for one-third of all land in sub-Saharan Africa (Earthtrends, 2009). These regions are important to investigate in more detail not only because of their size, but also because of the widespread poverty associated with the low productivity and mismanagement of natural resources in these regions. The vulnerability of these land types to soil erosion and climate change may even result in a worsening of this situation in the future (EIA, 1999). At the same time, arid and semi-arid regions have substantially different conditions and requirements for bioenergy production than more humid regions. Differences include current land use and land availability for bioenergy production, type of production system, yield potentials and the economics of production. Land use in arid and semi-arid regions is different from other regions due to lower population densities and because more land is used for grazing (as opposed to crop production) with lower livestock densities than in other regions. Moreover, lower water availability, lower crop productivity and different input requirements imply that different bioenergy crops must be considered in semi-arid and arid regions than in other regions. The economics of bioenergy production is also different than in more productive regions.

This study investigates the current technical and economic bioenergy production potential of three bioenergy production systems (cassava ethanol, jatropha oil and fuelwood) in semi-arid and arid sub-Saharan Africa. First, the land area that is available for bioenergy production is determined, accounting for other land uses such as biodiversity conservation and agricultural production and for the suitability of land for energy crop production. Next, the crop yields and production costs of these systems are estimated and cost-supply curves are constructed. The analysis focuses on various countries with large semi-arid and arid areas. In order to capture the variability in conditions found in different countries in the three geographical regions of sub-Saharan Africa, the following eight countries are assessed: Kenya and Tanzania (East Africa), Burkina Faso, Mali and Senegal (West Africa) and Botswana, South Africa and Zambia (Southern Africa).

The remainder of this paper is organized as follows. In Section 4.2, the methodology used for calculating technical and economic potentials is explained, and the bioenergy production chains are described. Section 4.3 explains the cost data used in the potential analyses. The results of the analyses on land availability and technical and economic potentials are presented in Section 4.4. Section 4.5 discusses the results, methodological choices and uncertainties in the data, and Section 4.6 concludes the paper with final remarks.

4.2 APPROACH

4.2.1 Technical potential

In this study, the current technical potential of bioenergy production in arid and semi-arid areas accounts for the current availability of land for bioenergy production and the bioenergy production system. Available land for bioenergy production in semi-arid and arid regions of the eight countries studied is defined as land that remains after current high biodiversity areas (including protected areas, biodiversity hotspots, forests and wetlands), agricultural land (including pastureland) and unsuitable areas (such as cities, deserts and steep slopes) are excluded. Section 4.2.1.1 describes these land categories and the method for assessing available land. The three bioenergy production systems (cassava ethanol, jatropha oil and fuelwood) that are assessed in this study are described in section 4.2.1.2, and methods for determining crop and energy yields are explained in section 4.2.1.3. Based on spatially explicit land availability and yield maps, the technical potential is determined by multiplying available land and yield per pixel, then summing it up for each region. The technical potential is expressed in petajoule (PJ) lower heating value.

4.2.1.1 Available land

Arid and semi-arid regions of the eight countries are demarcated by digitizing the World Meteorological Organization (WMO) and the United Nations Environment Programme's (UNEP) (WMO and UNEP, 2001) Aridity Zones map of Africa, in which arid and semi-arid regions are defined based on the aridity index (the ratio of mean annual precipitation to mean annual potential evapotranspiration). An aridity index between 0.05 and 0.2 designates arid regions, while an aridity index between 0.2 and 0.5 indicates semi-arid regions (WMO and UNEP, 2001). In a geographic information system (GIS) (ESRI's ArcGIS 9.3 software), raster datasets are compiled for the following land categories that are excluded from availability for bioenergy production.

- **Unsuitable areas** include cities, bare rock, sandy desert and dunes, stony deserts and water bodies as defined by the Global Land Cover 2000 database (GLC2000) (2003). Also excluded as unsuitable are areas with steep slopes in order to avoid exacerbating soil erosion through bioenergy production. Slopes steeper than 8% are excluded because of their limitation for agricultural crops and their increased erosion potential (Sys *et al.*, 1991). While slopes of up to 16% are still suitable for perennial crops, the 8% limit is applied here as a conservative estimate. Slopes are mapped with the IIASA and FAO (2000) median slope gradient map.
- **High biodiversity areas** comprise internationally and nationally protected areas as defined in the World Database on Protected Areas (WDPA) (UNEP-WCMC and IUCN WCPA, 2009), biodiversity hotspots (Conservation International, 2005) and forests and wetlands as defined by GLC2000 (2003). A *protected area* is understood here to be "a clearly defined geographical space, recognized, dedicated and managed, through legal or other effective means, to achieve the long-term conservation of nature with associated ecosystem services and cultural values" (Dudley, 2008). The WDPA includes world heritage sites, wetlands according to the Ramsar Convention, reserves that form part of the

UNESCO Man and the Biosphere Program and other internationally recognized sites, plus nationally designated sites (IUCN categories I – VI, not designated and no category) (UNEP-WCMC and IUCN WCPA, 2009). *Biodiversity hotspots* are characterized “both by exceptional levels of plant endemism and by serious levels of habitat loss” (Conservation International, 2005). Because of their generally high biodiversity levels, all categories of tree cover and wetlands as defined by GLC2000 (2003) are excluded from availability.

- **Agricultural land** consists of cultivated and managed areas, mosaic of cropland/tree cover/other natural vegetation and mosaic of cropland/shrubland and/or grass cover as defined by GLC2000 (2003) and pastureland (Ramankutty *et al.*, 2008). While GLC2000 defines *croplands* as “areas with over 50% cultures or pastures” (Mayaux *et al.*, 2003), pastures are specifically referred to as sown pastures and, as a result, do not account for natural grasslands and shrublands that are often used for grazing. Not accounting for natural grasslands and shrublands used for raising livestock would overestimate the actual availability of land for energy crop production. Therefore, this study also uses the pastureland map for 2000 from Ramankutty *et al.* (2008) to ensure grazing land is excluded from availability for energy crop production.

While the datasets refer to various years between 2000 and 2009, the results of the land availability analysis are considered valid for the present situation and for the near future. However, this may result in an underestimation of potentially available land areas because, while agricultural land areas in the eight countries have hardly changed since 2000 (FAOSTAT, 2009), decreases in forest and other wooded land have been observed. For example, in Tanzania deforestation rates were slightly higher than 1% per year between 2000 and 2005 while losses in other wooded land areas were nearly 15% per year in the same time period (FAO, 2006a).

4.2.1.2 Bioenergy systems for semi-arid and arid conditions

This study investigates three bioenergy production chains that may be promising for semi-arid and arid conditions in sub-Saharan Africa: cassava ethanol, jatropha oil and fuelwood. The reasons for choosing these bioenergy products and the crop cultivation, processing and energetic uses of each are described next.

Cassava ethanol

Cassava (*Manihot esculenta* Crantz) is chosen because of the existing experience with cassava production in sub-Saharan Africa, low skill and input requirements, its drought tolerance, potential production on marginal land and its suitability for bioethanol production (El-Sharkawy, 1993; Hu *et al.*, 2004; Nguyen *et al.*, 2008; Nadir *et al.*, 2009). Cassava ethanol is currently being produced in China and Thailand (Hu *et al.*, 2004; Nguyen *et al.*, 2008). The cassava-to-ethanol conversion process is a well established technology, and the conversion process in sub-Saharan Africa is assumed to be the same as in Asia.

Land preparation for cassava cultivation comprises ploughing and ridging and takes place before the wet season. Stem cuttings are prepared from the stems that are left after the roots are separated at harvest. The stems are planted manually. Manual weeding takes place during the first few months until the cassava plants develop shade large enough to compete for sunlight. Cassava is most commonly harvested manually eight months after planting at the earliest (Center for New Crops and Plant Products, 2009).

Fresh cassava roots contain approximately 30% starch (Hu *et al.*, 2004; Nguyen *et al.*, 2008). With a higher starch content, which can be obtained by chipping and drying the cassava roots, ethanol conversion efficiency can be improved (Atthasampunna *et al.*, 1987). The process for obtaining ethanol from cassava includes feedstock pretreatment (washing and crushing), pulp cooking, saccharification by either mixing the pulp with hydrochloric acid or sulphuric acid in pressure cookers or by partial hydrolysis and enzymatic treatment (transforming pulp into fermentable sugars), neutralization (removing the free acids and bringing the pH value in the range 5.0-7.0 to allow fermentation), fermentation, distillation and dehydration (Hu *et al.*, 2004; Nguyen *et al.*, 2008).

Jatropha oil

Jatropha (*Jatropha curcas* L.) is included in this analysis because it has been claimed by many to be drought tolerant (see, for example, Openshaw (2000) and van Eijck (2007)). If this is the case, jatropha could be promising for bioenergy production in semi-arid and arid regions. However, there is thus far little scientific evidence for this claim, and recent reports reveal much lower yields than originally posted (Achten *et al.*, 2008; Maes *et al.*, 2009). An additional reason for including jatropha is that the by-products of jatropha oil production, especially the press cake, have a high economic value. Press cake can be used as fertilizer, fuel in industrial boilers and for biogas and power generation (Achten *et al.*, 2008).

Jatropha is a large perennial shrub or small tree that produces seeds rich in oil and that can live to more than 50 years. The root system of jatropha plants consists of three to four lateral roots and a vertical taproot, which can reach five meters into the soil. The establishment of a jatropha plantation generally requires clearing land and preparing planting pits. Jatropha can be propagated through direct seeding or planting of stem cuttings. Seedlings are planted at the beginning of the rainy season to help develop a healthy taproot system (Achten *et al.*, 2008). Crop maintenance includes weeding, fertilization, pesticide application, pruning, thinning and clearing of firebreaks (Achten *et al.*, 2008). The shrub produces fruit between five months and three years after planting depending on the climatic and soil conditions. Harvesting begins approximately two to three months after the beginning of rainy season.

Approximately 34% of the (non-edible) jatropha seeds (by mass) are oil, which is assumed here to be extracted mechanically by, for example, an engine-driven screw press (Achten *et al.*, 2008). For the potential analysis in this study, jatropha oil is assumed to be used directly as a diesel substitute because transesterification is very expensive when methanol is not available locally and has to be imported at high costs, which is likely to be the case in the regions under consideration.

Fuelwood

Fuelwood from short rotation forestry plantations is included in this analysis because there are tree species that perform well in arid and semi-arid conditions (for example, *Acacia*, *Leucaena* and *Prosopis* species) (Nyadzi *et al.*, 2003a; Kimaro *et al.*, 2007; Wiskerke *et al.*, 2010). Furthermore, woody biomass can be directly used in the current energy systems as well as in more modern applications such as electricity and liquid fuels once they become more widely available. In addition, improvements in soil conditions (Buresh and Tian, 1997; Nyadzi *et al.*, 2003b) and low management and operation requirements after the initial establishment compared to annual energy crops are further reasons for including fuelwood production.

Short rotation forestry is the cultivation of fast growing hardwoods, planted at high density and generally harvested two to twelve years after planting. In the case of arid and semi-arid climates, the rotation period is likely longer than in more humid climates in order to allow for more efficient harvesting. Coppicing species are chosen so that new shoots emerge from the stump after the harvest and continue growing until the next harvest.

The establishment of a short rotation forestry plantation includes land clearing, ploughing and planting of seedlings. Operation and maintenance of a plantation generally includes weeding, fertilizer and pesticide application, as well as clearing of firebreaks. During harvest, the stems are cut down by chainsaw to near ground level. The harvested wood can be used for various energetic purposes such as fuel wood for cooking, heating and lighting; (co-)firing for electricity production; liquid fuel production via fermentation or gasification (Fischer/Tropsch process). In this study, the focus is placed on wood used as fuelwood because it does not require major modifications to current energy use and can, at the same time, help avoid some negative impacts of the current system such as long collection time and local deforestation.

4.2.1.3 Yields

Yield estimates for semi-arid and arid regions in sub-Saharan Africa are scarce in the literature, and those data points that were found vary significantly between regions and years (for yield estimates for cassava production see, for example, Tewe (2004), Central Statistical Office Zambia (2009), FAOSTAT (2009), Government of Burkina Faso (2009); for jatropha see, for example, Openshaw (2000), Jongschaap *et al.* (2007), Van Eijck (2007), Achten *et al.* (2008); and for woody biomass see, for example, Marrison and Larson (1996), Mead (2001), Nyadzi *et al.* (2003a), IPCC (2006), Kimaro *et al.* (2007)). As a result of large variations in yield data in the literature, this study applies the yields of the Crop and Grass Production of Model (CGPM) of the Integrated Model to Assess the Global Environment (IMAGE) (Leemans and Born, 1994; MNP, 2006). The CGPM generates global yield maps with a resolution of 0.5 degrees using soil and climate data. The use of the CGPM ensures that a consistent procedure is used for estimating yields for the three cropping systems and all eight countries and allows for a differentiation of yields for arid and semi-arid regions.

For this study, the CGPM maps for cassava are calibrated assuming an average yield of 4.8 t ha^{-1} y^{-1} of fresh cassava roots in semi-arid sub-Saharan Africa (Sarma and Kunchai Darunee, 1991). This calibration is carried out in order to better account for the generally lower yields in sub-Saharan Africa as a result of the manual labour-based and low-input management system of cassava production in this region.

For jatropha, no yield maps exist from the CGPM model. To allow a spatial differentiation of jatropha seed yields, the CGPM oil crop yield map is calibrated for jatropha based on an average seed yield in semi-arid regions of 2.5 t ha^{-1} y^{-1} (Achten *et al.*, 2008). This figure is in line with (the few) other values found in the literature for semi-arid regions. For example, Wiskerke *et al.* (2010), considered a yield of 2.4 t ha^{-1} y^{-1} for semi-arid Shinyanga in Tanzania.

The CGPM woody crop yield map is calibrated by multiplying the theoretical yields by a management factor of 0.7, which represents the gap between theoretically feasible crop yields and actual crop yields resulting from lower-than-optimal management and from harvest losses (Hoogwijk *et al.*, 2005). This results in an average biomass yield of 7.4 t dry matter (t dm) ha^{-1} y^{-1} for semi-arid regions.

4.2.2 Economic potential

The economic potential is the part of the technical potential that can be produced at economically profitable levels (Hoogwijk *et al.*, 2005). In this study, the economic potential is determined by constructing cost-supply curves for cassava ethanol, jatropha oil and fuelwood production in semi-arid and arid regions of the eight countries investigated. These curves are made by ranking the geographic potential as a function of production costs per grid cell. The production costs (in US\$ GJ^{-1}) are calculated by applying the discounted value for biomass yields and production costs because production costs and benefits from biomass harvest are distributed unequally over time (Van den Broek *et al.*, 2000). The costs of the main components of feedstock production (land, labour and inputs), transportation costs from plantation to the conversion plant and conversion costs are assessed from the literature. In order to allow a more meaningful comparison with reference energy prices, taxes, wholesale margins, retail margins and distribution costs are estimated based on values from the literature. The fertilizer requirements are determined by means of a nutrient balance methodology, which assumes that the nutrients taken up by the crop during its growth must be replenished by fertilizers in order to maintain the soil's nutrient composition (De Wit and Faaij, 2010). While this is a simplification of actual practice, it enables a fair comparison of fertilizer requirements in different regions with different productivities.

Transportation costs are based on unit costs of transport by truck (US\$ km^{-1} t^{-1}) and transportation distances. While transport costs are determined from literature, the transport distance from the field to the processing unit is estimated on the basis of the "delivery area" around the pre-treatment plant (Perlack and Turhollow, 2003), which depends on the coverage of the energy crop of the total area (as determined in the land availability analysis), the yield (as determined in the technical potential analysis) and the plant capacity (determined from the literature). Assuming that the processing plant is located at the centre of a circle, the average transportation distance is taken as the radius of the circle multiplied by a road winding factor of 1.3 that accounts for the actual road distance rather than a straight line distance (Perlack and Turhollow, 2003).

4.3 INPUT DATA

The datasets used in the land availability analysis are described in Section 4.2.1.1. An overview of input data for the technical and economic potential analysis is shown in Table 4.1 and Table 4.2.

Table 4.1: Overview of average yields, wages, land costs, fertilizer costs, transport distances and transportation costs in eight sub-Saharan African countries

	Botswana	**Burkina Faso**	**Kenya**	**Mali**	**Senegal**	**South Africa**	**Tanzania**	**Zambia**
Average yields in semi-arid / arid regions (t ha^{-1} y^{-1}) [a]								
- Cassava roots	2.3 / 0.2	3.8 / 1.8	7.5 / 4.4	3.4 / 0.6	2.8 / 1.2	4.8 / 0.4	8.9 / n.a.	4.9 / n.a.
- Jatropha seeds	2.5 / 0.2	3.1 / 2.4	2.4 / 2.0	2.6 / 0.7	2.7 / 1.7	2.3 / 0.3	2.5 / n.a.	2.7 / n.a.
- Fuelwood	5.5 / 0.7	10.0 / 6.3	12.4 / 8.9	8.1 / 2.7	7.4 / 5.7	8.7 / 1.1	12.4 / n.a.	9.5 / n.a.
Wage [b] (US\$ h^{-1})	1.0	0.4	0.3	0.4	0.4	4.0	0.3	0.3
Land costs [c] (US\$ ha^{-1} y^{-1})	93	22	20	22	22	93	20	20
Fertilizer costs [d]								
- Urea (US\$ (t N)$^{-1}$)	1000	635	494	635	635	500	529	1000
- Single super-phosphate (US\$ (t P_2O_5)$^{-1}$)	581	1004	1004	1004	1004	581	1004	581
- Mutriate over 45% K_2O (US\$ (t K_2O)$^{-1}$)	521	693	500	693	693	440	693	521
Transport distance semi-arid / arid regions (km)								
- Cassava [e]	39 / 121	40 / 164	26 / 35	37 / 92	57 / 40	69 / 636	35 / n.a.	36 / n.a.
- Jatropha and fuelwood [e]	6 / 16	7 / 21	7 / 8	6 / 12	9 / 5	15 / 115	10 / n.a	7 / n.a
Transportation costs [f] (US\$ t^{-1} km^{-1})	0.05	0.08	0.07	0.08	0.08	0.05	0.07	0.07

n.a. – not applicable

a – MNP (2006): Area-weighted average yields are determined based on the yield maps (Section 4.2.1.3) and the delineation of semi-arid and arid regions according to the WMO and UNEP (2001). The average refers to the whole arid or semi-arid region and does not exclude areas that are marked unavailable or unsuitable in the land analysis.

b – ILO (2009); wages in Tanzania are assumed to be representative of other East African countries for which no data is available; West African countries are assumed to have slightly higher wages than East African countries because of generally higher GDP in West Africa; South African wages are given for the mining sector, it is here assumed that wages in agriculture are only two-thirds of those in the mining sector;

c – Hoogwijk *et al.* (2009) present land costs for global regions, including East, West and Southern Africa. Their data is applied here due to lack of country (or even sub-national) specific land costs;

d – FAOSTAT (2009); **N**: urea prices paid by farmers - as urea prices for Mali and Senegal are not given, they are assumed to have the same price as neighbouring Burkina Faso; **P**: single superphosphate prices paid by farmers - only available for Kenya and South Africa. East and West Africa countries are assumed to have the same price as Kenya, while Botswana and Zambia are assumed to be similar to South Africa; **K**: mutriate over 45% K_2O prices paid by farmers - data available only for Tanzania, South Africa, Kenya and Botswana; Burkina Faso, Mali and Senegal are assumed to be similar to Tanzania and prices in Zambia are assumed to similar to Botswana.

e – Transport distances are calculated as described in Section 4.2.2 applying a plant capacity for jatropha oil of 1.5 m^3 per day and for cassava ethanol of 100 m^3 per day; transport distances for jatropha seeds are used for fuelwood transport distances because fuelwood is assumed to be consumed locally;

f – World Bank (2009); transportation costs are given for East, West and Southern Africa and assumed to be representative for arid and semi-arid regions of the eight countries studied here despite infrastructure generally being worse in these regions;

Table 4.2: Overview of average labour requirements, fertilizer requirements and conversion costs for cassava, jatropha and fuelwood

	Cassava	**Jatropha**	**Fuelwood**
Average annual labour requirements ($h\ ha^{-1}\ y^{-1}$)	530 [a]	500 [b]	240 [c]
Fertilizer requirements [d]	(kg N / P / K $(t\ dm)^{-1}$)	(kg N / P / K $(t\ seeds)^{-1}$)	(kg N / P / K $(t\ dm)^{-1}$)
Nitrogen	6.4	3.1	6.7
Phosphor	1.9	0.1	1.1
Potassium	7.9	0.9	2.3
Conversion costs (US$ GJ^{-1})	8.9 [e]	5.4 [f]	-

a – Econergy International Corporation (2008); labour requirement for cassava production in Mozambique;

b – Jongschaap *et al.* (2007) assume labour requirements in the first year of 22 person days $ha^{-1}\ y^{-1}$ and an increase to 70 person days $ha^{-1}\ y^{-1}$ in the sixth year. Assuming then that from year 6 to year 20, 70 person days $ha^{-1}\ y^{-1}$ are required, the annual average is 63 person days $ha^{-1}\ y^{-1}$ or 500 $h\ ha^{-1}\ y^{-1}$;

c – Wiskerke *et al.* (2010); labour requirements for manual based fuelwood production in semi-arid Shinyanga, Tanzania;

d – Fertilizer requirements are calculated as described in Section 4.2.2 assuming crop specific nutrient composition of 4.50 kg N $(t\ dm)^{-1}$, 0.83 kg P $(t\ dm)^{-1}$ and 6.6 kg K $(t\ dm)^{-1}$ for cassava (Howeler, 2002); 2.2 kg N $(t\ seeds)^{-1}$, 0.05 kg P $(t\ seeds)^{-1}$ and 0.73 kg K $(t\ seeds)^{-1}$ for jatropha (Jongschaap *et al.*, 2007) and 2.57 kg N $(t\ fresh\ weight)^{-1}$, 0.26 kg P $(t\ fresh\ weight)^{-1}$ and 1.05 kg K $(t\ fresh\ weight)^{-1}$ for fuelwood (Singh *et al.*, 1997); fertilizer factor for nitrogen of 1 kg N $(kg\ N)^{-1}$, for phosphor of 2.3 kg P_2O_5 $(kg\ P)^{-1}$ and for potassium of 1.2 kg K_2O $(kg\ K)^{-1}$ (De Wit and Faaij, 2010); nitrogen uptake factor of 60% (De Wit and Faaij, 2010);

e – Nguyen *et al.* (2008); it is assumed that cassava ethanol conversion costs in sub-Saharan Africa are comparable to those in Thailand;

f – Openshaw (2000); seed processing and oil manufacturing costs for motor press.

Cassava ethanol conversion costs of 210 US\$ m^{-3} are taken from production in Thailand (Nguyen *et al.*, 2008) because it is assumed that the conversion process and its costs are similar in sub-Saharan Africa. Cassava-to-ethanol conversion efficiencies are given in Gibbs *et al.* (2008) as 180 l t^{-1} fresh roots, in Jansson *et al.* (2009) as 150 l t^{-1} fresh roots and in Econergy International Corporation (2008) as 200 l t^{-1} fresh roots. Atthasampunna *et al.* (1987) determine conversion efficiencies to be between 185 and 200 l t^{-1} in laboratory conditions, depending on the starch content. In this study, Jansson *et al.*'s conversion efficiency is applied as a conservative estimate. Furthermore, an ethanol density of 800 kg m^{-3} and an ethanol energy content of 26.4 MJ kg^{-1} are applied (Girard and Fallot, 2006).

Jatropha oil conversion costs of 200 US\$ m^{-3} proposed by Openshaw (2000) are applied here. Jatropha oil extraction rates vary for different methods of extraction (Achten *et al.*, 2008). This study applies an average oil extraction rate for mechanical extraction of 75% (Achten *et al.*, 2008). An oil content of jatropha seeds of 34%, an oil density of 900 kg m^{-3} and an energy content of jatropha oil of 40.7 MJ kg^{-1} are applied (Wiskerke *et al.*, 2010). A jatropha plantation lifetime of 20 years, from which jatropha can be harvested annually from the third year onwards, is assumed. A discount rate of 10% is applied for determining the discounted production costs of jatropha seeds.

Taxes, wholesale margins, retail margins and distribution costs for the eight countries are estimated by applying the percentage share of these costs in conventional gasoline and diesel prices of each country as a proxy (ERB, 2008; Kojima *et al.*, 2010).

Fuelwood is assumed to be consumed locally, and therefore the same average transport distances as determined for jatropha are assumed. Fuelwood production costs are calculated for a plantation lifetime of 21 years, assuming that harvesting takes place every seven years. An energy content of fuelwood of 20 MJ kg^{-1} and a discount rate of 10% are applied.

4.4 RESULTS

4.4.1 Available land

As an example of the location of the different land categories that are excluded from availability, Figure 4.1 depicts the different land categories in the case of Tanzania. Similar maps are also generated for the other countries. The combination of these maps results in the location of available and unavailable land areas in the semi-arid and arid regions of the eight countries as depicted in Figure 4.2.

Table 4.3 presents an overview of land resources in the arid and semi-arid regions of the eight countries. Available land ranges from less than 2% of the total semi-arid and arid land area in South Africa to 21% in Botswana; in absolute terms, land availability ranges from 1.2 Mha in Senegal to 12.2 Mha in Botswana (Table 4.3). Table 4.3 indicates that in most countries the most important limiting factor for bioenergy production in land availability is agricultural land. While in Burkina Faso and Senegal cropland is more important, in all other countries pastureland affects land availability the most. In Kenya, South Africa and Tanzania, steep slopes and biodiversity hotspots are also important aspects that reduce land availability.

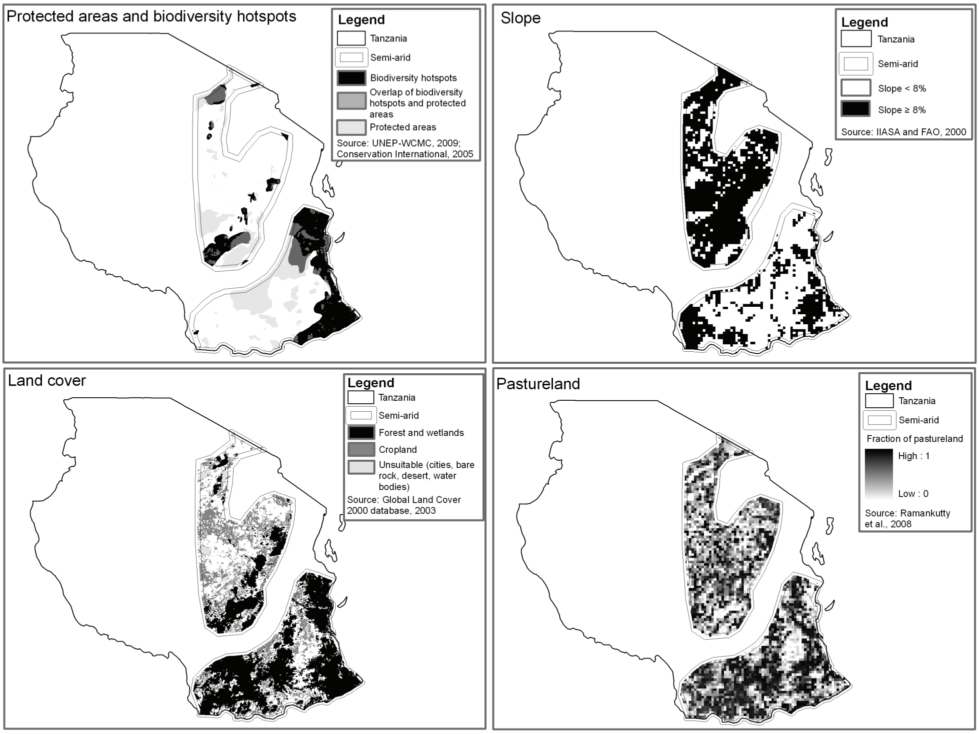
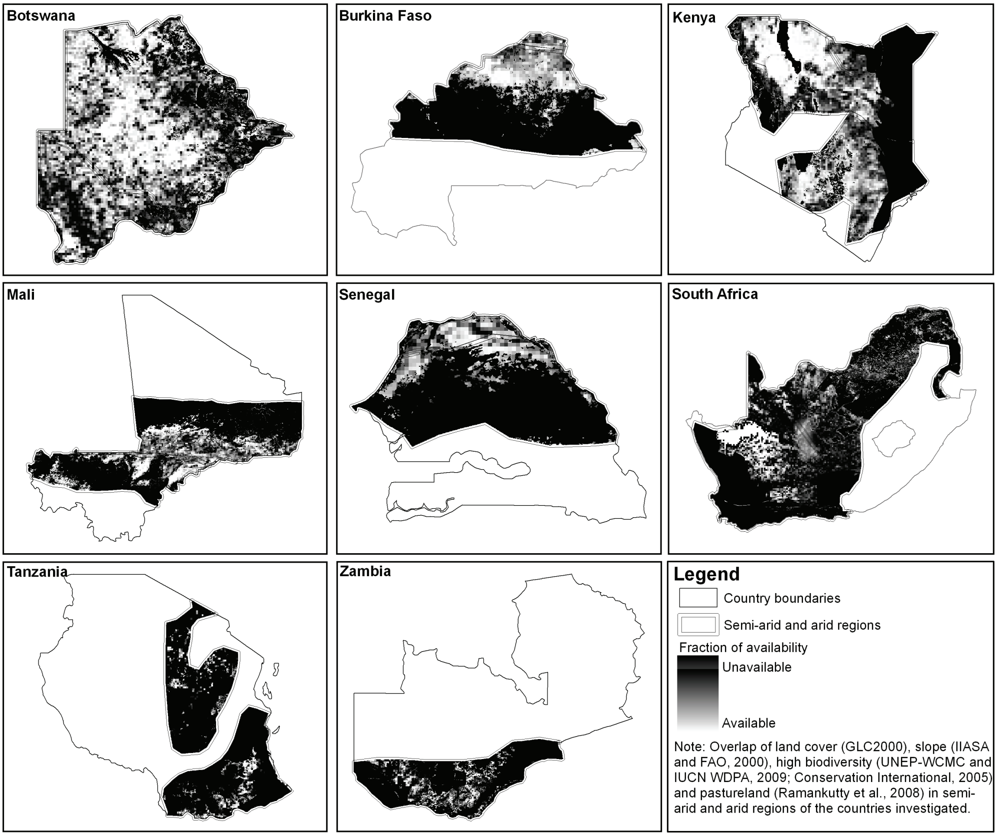


Figure 4.1: Maps of protected areas and biodiversity hotspots, slope, land cover and pastureland in Tanzania in 2000

4.4.2 Technical potential bioenergy production

The technical potential of fuelwood is significantly higher than cassava ethanol and jatropha oil (Table 4.4). This is a result of high yields and higher primary energy content of fuelwood than cassava roots or jatropha seeds. While a direct comparison of the technical potential of fuelwood with that of cassava ethanol or jatropha oil is not meaningful due to the different end products of the three cropping systems, it is clear that even if losses for converting fuelwood to electricity or liquid fuel are accounted for, these energy carriers would have a higher technical potential than cassava ethanol or jatropha oil (Table 4.4). Despite the lower potential of cassava ethanol and jatropha oil, all three crops can significantly contribute to current energy demand in the eight countries (Table 4.4). The fuelwood potential in Botswana, Kenya, Senegal and Zambia is higher than the current estimated consumption of combustibles, renewables and waste, which is mostly traditional biomass in sub-Saharan Africa. The cassava ethanol and jatropha oil potentials are comparable to the current consumption of petroleum products in most countries. Even when considering that energy consumption has been increasing over time (*e.g.* ranging from 1% per year in Botswana to 4% per year in Tanzania between 2000 and 2006

(OECD/IEA, 2009)), bioenergy can play a significant role in future provision of energy in these countries.



Note: Overlap of land cover (GLC2000), slope (IIASA and FAO, 2000), high biodiversity (UNEP-WCMC and IUCN WDPA, 2009; Conservation International, 2005) and pastureland (Ramankutty et al., 2008) in semi-arid and arid regions of the countries investigated.

Figure 4.2: Available land for bioenergy production in Botswana, Burkina Faso, Kenya, Mali, Senegal, South Africa, Tanzania and Zambia in 2000

4.4.3 Economic potential of bioenergy production

The average production costs of cassava ethanol, jatropha oil and fuelwood in semi-arid and arid regions of the eight countries and the national market prices of the reference products are depicted in Figure 4.3. Cassava ethanol production costs range from 36 US\$ GJ^{-1} in semi-arid Tanzania and Kenya to 1,836 US\$ GJ^{-1} in arid South Africa. Jatropha oil production costs range from 26 US\$ GJ^{-1} in semi-arid Zambia, Tanzania, Kenya and Burkina Faso to 889 US\$ GJ^{-1} in arid South Africa. Fuelwood production costs range from 1.9 US\$ GJ^{-1} in semi-arid Kenya to 23.8 US\$ GJ^{-1} in arid Botswana. In all three production systems, the production costs are always lower in semi-arid than in arid regions within a country, which is mainly the result of the higher yields in semi-arid regions. The higher production costs in Botswana and South Africa compared to other sub-Saharan African countries are due to the labour intensive production systems that are considered in combination with the higher wages in Botswana and South Africa (see also Section 4.5).

Table 4.3: Availability of land for energy crop production in semi-arid and arid regions of eight sub-Saharan African countries

	Botswana	**Burkina Faso**	**Kenya**	**Mali**	**Senegal**	**South Africa**	**Tanzania**	**Zambia**
Total area (Mha)	**57.6**	**27.2**	**58.0**	**124.9**	**19.5**	**121.3**	**93.3**	**75.0**
- Semi-arid area	44.9	14.3	22.3	24.5	9.7	37.6	31.5	16.0
- Arid area	12.8	0.5	23.1	39.3	1.5	51.4	0.0	0.0
- Other areas	0.0	12.4	12.6	61.1	8.3	32.3	61.8	59.0
Excluded area[a] (Mha)	**45.4**	**13.2**	**39.8**	**54.9**	**10.0**	**87.7**	**29.6**	**14.3**
- Unsuitable								
-- Cities, bare rock, sandy desert and dunes, stony deserts and water bodies	0.7	0.0	0.8	21.6	0.2	1.0	0.3	0.2
-- Steep slopes (≥ 8%)	6.0	3.0	19.3	8.4	0.8	62.6	14.5	5.0
- High biodiversity								
-- Protected areas	18.7	1.8	3.1	2.7	2.8	5.8	6.4	5.3
-- Biodiversity hotspots	0.0	0.0	11.5	0.0	0.0	17.4	6.0	0.0
-- Closed canopy forest and wetlands	3.6	0.0	2.2	0.1	0.0	7.4	11.5	4.6
- Agricultural land								
-- Cropland	4.1	8.9	2.6	14.4	7.1	14.3	4.7	2.4
-- Pastureland	21.4	3.1	15.6	20.1	3.1	47.8	13.3	5.3
Available area (Mha)	**12.2**	**1.6**	**5.6**	**8.9**	**1.2**	**1.3**	**1.9**	**1.7**
- Semi-arid	8.4	1.6	2.8	3.4	0.7	1.1	1.9	1.7
- Arid	3.8	0.0	2.8	5.5	0.5	0.2	0.0	0.0
Share of available area in total arid and semi-arid area (%)	**21.2**	**10.7**	**12.4**	**14.0**	**10.8**	**1.5**	**6.1**	**10.6**

a – The sum of the different land areas excluded from availability does not equal the total excluded area because of overlaps between different categories. This breakdown is presented here to give insight into the importance of the different land categories excluded.

The average production costs of cassava ethanol are in most regions dominated by the feedstock costs, but conversion costs and taxes also play an important role. The production costs of jatropha oil in most regions are dominated by feedstock production and other costs (taxes, wholesale margin, retail margins and distribution costs), while conversion costs are less important. Conversion costs of both cassava ethanol and jatropha oil comprise smaller shares in the total production costs in regions with low yields, mainly arid areas.

A comparison of the average production costs to the market prices of the reference products indicates that cassava ethanol production cannot compete with gasoline in any of the regions analyzed, though it comes close in semi-arid Tanzania, Kenya and Zambia. Jatropha oil can be produced at average costs lower than market prices for diesel only in semi-arid Zambia, but comes close to the market price in semi-arid Burkina Faso, Senegal and Tanzania. Production costs of fuelwood are comparable to the average market price of fuelwood only in semi-arid Tanzania and Kenya and slightly higher in semi-arid Zambia, Burkina Faso, Mali and Senegal and arid Kenya.

Table 4.4: Overview of technical potential in semi-arid and arid regions in sub-Saharan Africa

	Botswana	**Burkina Faso**	**Kenya**	**Mali**	**Senegal**	**South Africa**	**Tanzania**	**Zambia**	**Total**
Cassava ethanol total (PJ y^{-1})	52	15	103	46	6	10	55	25	313
- arid	3	0	40	17	2	0	0	0	62
- semi-arid	49	15	62	29	4	10	55	25	250
Jatropha oil total (PJ y^{-1})	202	52	76	166	26	25	50	48	645
- arid	12	0	69	74	9	1	0	0	164
- semi-arid	190	52	7	92	18	24	50	48	481
Fuelwood total (PJ y^{-1})	827	277	1141	977	152	118	459	315	4265
- arid	70	1	464	479	56	6	0	0	1075
- semi-arid	757	276	677	498	95	113	459	315	3190
Total final consumption of energy in 2006 [b] (PJ y^{-1})	**64**	**No data**	**495**	**No data**	**70**	**2648**	**638**	**241**	
- coal and peat	6		3		4	604	1	4	
- petroleum products	29		118		31	814	54	24	
- gas	0		0		0	95	3	0	
- combustibles, renewables and waste	19		356		29	417	571	182	
- electricity	9		19		6	717	10	30	

a - South Africa considers jatropha an invasive species, which is why it is prohibited from cultivation there (Von Maltitz and Brent, 2009). However, there are businesses attempting to get this government decision reversed so that jatropha could be used for bioenergy production. The present study includes a calculation of the jatropha oil production potential in South Africa in order to indicate its potential there. However, any part of this potential is only realizable if the South African government reverses its decision on prohibiting jatropha cultivation.

b – OECD/IEA (2009)

The cost-supply curves for cassava ethanol, jatropha oil and fuelwood production in semi-arid and arid regions (Figure 4.4) indicate that in all countries more fuelwood can be produced at lower costs than bioenergy from the other two systems and that the economic potential of jatropha oil is generally higher than that of cassava ethanol. Based on the higher production costs and lower potentials in arid regions, the economic potential of arid regions is lower than that of semi-arid regions. Nevertheless, for example, in arid Kenya 270 PJ could be produced annually with fuelwood at production costs of less than 3 US\$ GJ^{-1}, which is competitive with coal market prices. In semi-arid regions the potential of fuelwood production at costs lower than 3 US\$ GJ^{-1} ranges from 82 PJ y^{-1} in Senegal to 456 PJ y^{-1} in Tanzania.

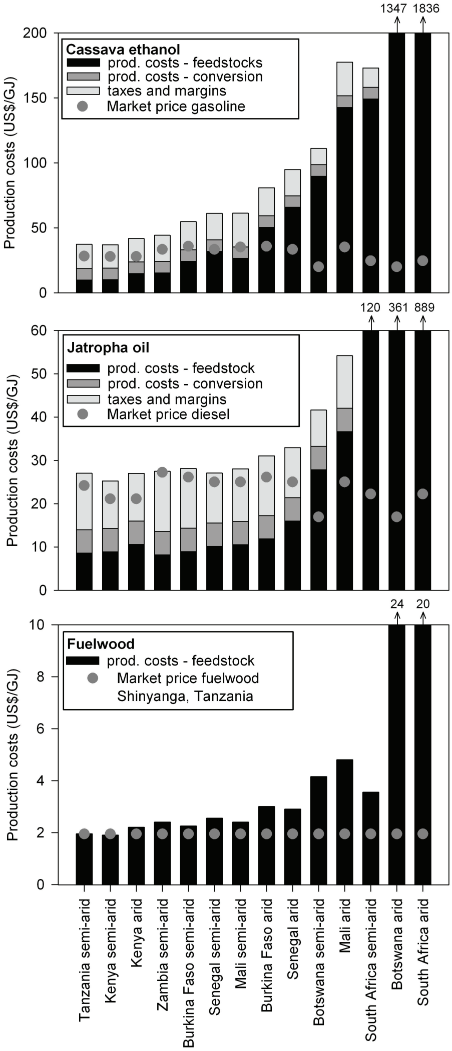


Figure 4.3: Production costs of cassava ethanol, jatropha oil and fuelwood in semi-arid and arid regions of eight sub-Saharan African countries compared to market prices of reference energy systems

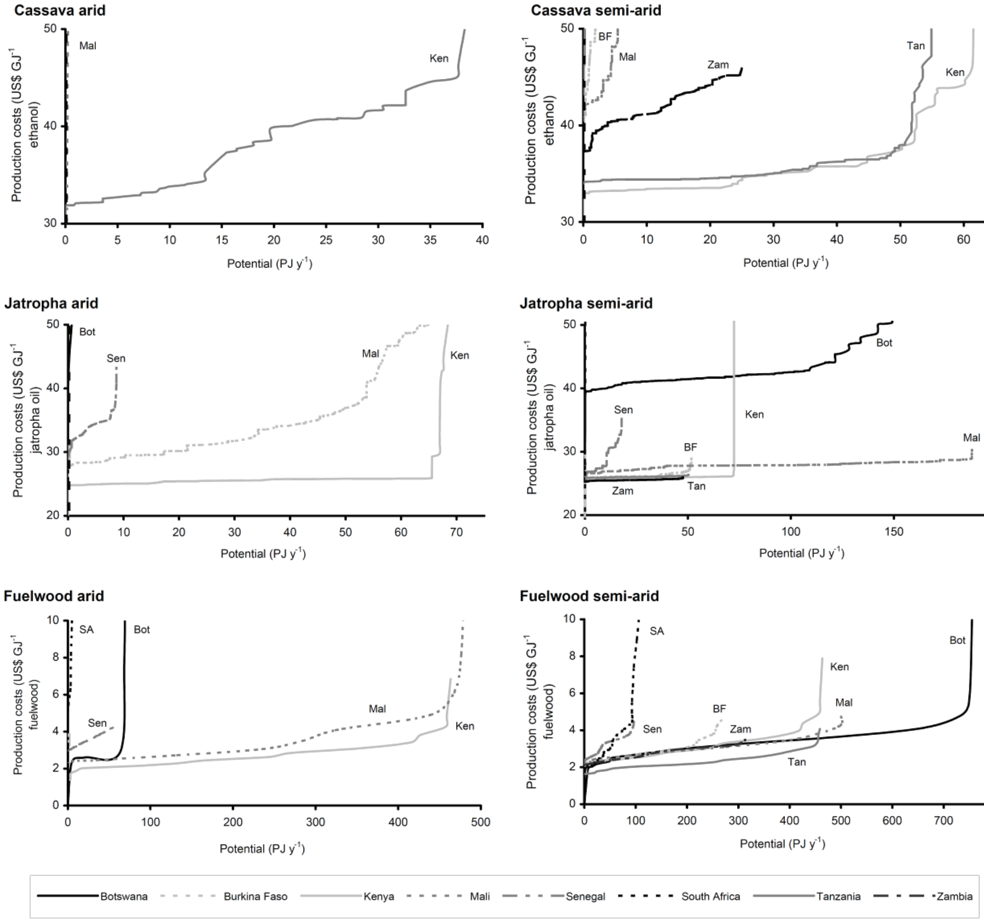


Figure 4.4: Cost-supply curves for cassava ethanol, jatropha oil and fuelwood production in semi-arid and arid regions of Botswana (Bot), Burkina Faso (BF), Kenya (Ken), Mali (Mal), Senegal (Sen), South Africa (SA), Tanzania (Tan) and Zambia (Zam)

4.5 DISCUSSION

Available land for bioenergy production has been determined by excluding land that is at present already used for other purposes, such as agriculture and nature conservation, or that is unsuitable for bioenergy production. While this study has attempted to exclude land based on the main competing uses, the datasets applied cannot account for all uses. An example of land uses that are particularly difficult to accurately capture are the use of land for livestock grazing, hunting and gathering and cultural services provided by land (Watson, 2009c). In this study, grazing is accounted for by applying data from Ramankutty *et al.* (2008), who provide a homogenous dataset of the world’s pasturelands applying the best available global data. However, this dataset possesses several uncertainties resulting from different definitions of pastures and grazing land, low quality of the census data of pasturelands (particularly in Africa), seasonal and inter-annual variability in grazing in semi-arid and arid regions due to potentially large variations in climates and to multiple uses within a year and, related to the latter, the intensity of grazing (see Ramankutty *et al.* (2008) for a detailed description of these shortcomings). However, these uncertainties and the effect on land availability for energy crop production could not be assessed due to a lack of quantitative data.

This study did not account for hunting and gathering or cultural services due to a lack of an appropriate methodology and data for spatially explicit mapping of these activities and services. In addition, Watson (2009a) has identified some localities in semi-arid and arid regions that should be avoided due to social constraints such as archaeological sites, areas that have cultural significance, areas with a long history of conflict over resources and areas destined for land reform. Not accounting for these areas as well as hunting and gathering or cultural services in the land availability analysis may result in bioenergy production displacing these forms of land use, which is likely to have negative social consequences. Considering that such displacement is unsustainable, future research needs to assess these uses and services and investigate how they affect land availability for bioenergy production. In addition, during the planning of actual projects an assessment of current land use and land ownership needs to be conducted in order to avoid any land use change and resulting conflicts with the local population.

While this study has analyzed the current situation, it is important to recognize that future demand for food and feed production as a result of population growth and dietary changes and/or stimulating domestic production may require additional land. Under the assumption that bioenergy should not compete with food production, this may reduce availability of land for bioenergy production. That said, the integration of food, feed and energy production through, for example, intercropping, rotational woodlots or hedgerows may provide an opportunity to minimize the competition for land and negative impacts associated with land use change. In addition, other studies (i.e. Hoogwijk *et al.*, 2005; Smeets *et al.*, 2007; Dornburg *et al.*, 2008) have shown that the potential for increasing food production efficiency is (very) high, especially in sub-Saharan Africa, so that increased food production does not (necessarily) have to result in lower land availability for energy crop production.

The land availability analysis accounted for high biodiversity areas by excluding protected areas, biodiversity hotspots and closed forests and wetlands. However, due to the underrepresentation of certain regions (including Africa) and ecosystems in the WDPA (UNEP-WCMC and IUCN WCPA, 2009), unavailability of other GIS-based data on biodiversity and lack of national and sub-national data on biodiversity in the eight countries investigated here, there may be other areas with high or unique biodiversity that have not yet been covered. Furthermore, Watson (2008) notes that protected areas in Africa only contain a limited, biased sample of biodiversity. This is mainly due to the fact that that protected areas in the region received their status either because they were unsuitable for commercial agriculture or because they served as a buffer between land claimed by white colonists and land allocated for native communities (Watson, 2008). In addition, the example of South Africa shows that very high biodiversity and endemism are also found outside of protected areas in semi-arid and arid regions (Watson, 2008). Examples of other ways of categorizing high biodiversity areas are 1) the Integrated Biodiversity Assessment Tool (2008), where biodiversity is mapped by global datasets for protected areas, biodiversity hotspots, key biodiversity areas, biodiversity conservation sites identified by the Alliance for Zero Extinction, high biodiversity wilderness areas and endemic bird areas, and 2) the UNEP-WCMC (2008) project on "Carbon and Biodiversity: A demonstration atlas", where six different global conservation prioritization schemes are combined. Areas with the largest number of overlaps are those considered to have the greatest degree of consensus on their importance for conservation/biodiversity. Future studies including such datasets as applied by the Integrated Biodiversity Assessment Tool (2008) or by UNEP-WCMC (2008) may improve the analysis but, even more important is the collection of detailed national-level datasets on biodiversity (M. Bertzky, UNEP-WCMC, personal communication). In addition to better mapping of high biodiversity areas and excluding those from land availability for bioenergy production, changing biodiversity levels as a result of converting different land types to bioenergy production also need to be assessed for the semi-arid and arid regions investigated here.

Uncertainties also exist in yield data. This study estimated the yields based on the results of the IMAGE model (Leemans and Born, 1994) because it allows for a consistent method for all eight countries and a differentiation of yields for arid and semi-arid regions. However, the calibrations for the cassava and jatropha yield maps in particular result in uncertainties. This study applied the average yield for semi-arid and arid regions in sub-Saharan Africa of 4.8 t ha^{-1} y^{-1} as suggested by Sama and Kunchain Darunee (1991) for the calibrations because it results in yields in semi-arid and arid regions that are lower than national average yields (FAOSTAT, 2009). Gibbs *et al.* (2008) have proposed higher yields (7.7 t ha^{-1} y^{-1}), but applying Gibbs *et al.*'s average yield would result in yields of semi-arid and arid regions which seem unrealistically high compared to the national average yields reported in FAOSTAT (2009). Still, future cassava yields may be even higher as there are many initiatives aimed at increasing cassava yields (including high-yielding varieties for drier conditions). For example, the USAID-funded project "Unleashing the power of cassava in response to the food price crisis" aims at yield improvements from current 7 and 12 t ha^{-1} y^{-1} across Nigeria, DR Congo, Ghana, Malawi, Mozambique, Sierra Leone and Tanzania to between 12 and 30 t ha^{-1} y^{-1} (Africa News, 2009).

For jatropha, this study calibrated the general oil crop yield map to estimate jatropha yields since no jatropha yield maps are currently available and since large ranges in yields are found in the literature (Openshaw, 2000; Jongschaap *et al.*, 2007; Van Eijck, 2007). This approach has the disadvantage of potentially underestimating jatropha yields because other oil crops may be more sensitive to dry conditions. However, since jatropha yields are still highly uncertain, this conservative estimation of yields was applied in this study. Bioenergy yields in arid and semi-arid conditions need to be assessed in more detail in the future, and research into improving yields should focus on management and the choice of crop variety suitable to arid and semi-arid conditions.

Production costs were determined based on existing information from the literature. However, the literature on cost data for bioenergy production in sub-Saharan countries is scarce (especially for cassava), meaning that various assumptions had to be made. For example, manual labour-based production systems are applied in this study for all three crops because of the possibility of generating jobs and the reduced access to and high prices of machinery, especially in remote areas of sub-Saharan Africa. In Botswana and South Africa, where wages are significantly higher than in other sub-Saharan African countries, this manual labour-based production system results in very high bioenergy production costs. In these countries, as well as in countries with low labour capacity, a more mechanized production system may reduce production costs. Future work needs to investigate the potential for mechanization of the three bioenergy systems assessed here and the impact on production costs.

In the absence of sub-national data, this study compared production costs to country average market prices for gasoline and diesel. Country averages, however, may be lower than prices in remote areas, such as many semi-arid and arid regions. As a result, bioenergy production may be economically interesting in even more regions than found in this study.

This study did not account for possible political risks, such as corruption, weak governance and political instability, associated with large investments in sub-Saharan Africa. These aspects must be considered when assessing the implementation of bioenergy production in sub-Saharan Africa.

This study applied land use sustainability criteria for bioenergy production, but there are many other criteria that need to be fulfilled before bioenergy production in semi-arid and arid regions in sub-Saharan Africa can be considered sustainable. Most important are environmental impacts (such as greenhouse gas emissions from land conversion, changes in soil conditions and soil fertility, and water use by the energy crops and its effects on local and regional hydrology systems/basins and local populations) and bioenergy production's social and socio-economic impacts (such as possible displacement of hunter/gatherer and nomadic pastoralist communities).

4.6 CONCLUSIONS

This study assesses the current technical and economic potential of three bioenergy production systems (cassava ethanol, jatropha oil and fuelwood) in semi-arid and arid sub-Saharan Africa. Results are presented for eight countries: Botswana, Burkina Faso, Kenya, Mali, Senegal, South Africa, Tanzania and Zambia. The results indicate that the availability of land for energy production ranges from 2% (1.3 Mha) of the total semi-arid and arid area in South Africa to 21% (12 Mha) in Botswana. Land availability for bioenergy production is restricted mainly by agricultural land use, but also by steep slopes and biodiversity protection. This study finds that the current technical and economic bioenergy potentials for cassava ethanol, jatropha oil and fuelwood production for semi-arid and arid regions in eight sub-Saharan African countries are high compared to the current final energy consumption in these countries. The current total technical potential for the semi-arid and arid regions of the eight countries is calculated to be approximately 300 PJ y^{-1} for cassava ethanol production, 600 PJ y^{-1} for jatropha biodiesel or 4,000 PJ y^{-1} for fuelwood. The potential contribution to current final energy consumption is lowest for cassava ethanol in South Africa (0.4%) and highest for fuelwood in Botswana (many hundred percent).

While the production costs of bioenergy vary significantly among the countries and regions investigated, fuelwood can be produced at costs similar to market price in many, especially semi-arid, regions. Cassava ethanol and jatropha diesel production can currently compete with conventional energy carriers in only few semi-arid regions. However, fuelwood, jatropha oil, and cassava ethanol production costs in most arid regions of sub-Saharan Africa are often above average national market prices of gasoline, diesel, and fuelwood. Despite high production costs, sustainable bioenergy production in semi-arid and arid regions of sub-Saharan Africa may still be desirable because it is a potential driver of rural economic and social development. Potential developmental benefits include allowing rural areas to produce more energy locally, increasing the availability of and access to energy, creating additional markets for agricultural products and helping generate more income for rural populations. Additional potential environmental benefits of perennial bioenergy crops include improving soil conditions, increasing soil carbon storage, reducing soil erosion and increasing agricultural productivity. These potential benefits can provide important (additional) reasons for further investigating and investing in sustainable bioenergy production in semi-arid and arid regions of sub-Saharan Africa.

4.7 ACKNOWLEDGEMENTS

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Chapter 5

The global technical and economic potential of bioenergy from salt-affected soils

Abstract: This chapter assesses the extent and location of salt-affected soils worldwide and their current land use/cover as well as the current technical and economic potential of biomass production from forestry plantations on these soils (biosaline forestry). The global extent of salt-affected land amounts to approximately 1.1 Gha, of which 14% is classified as forest, wetlands or (inter)nationally protected areas and is considered unavailable for biomass production because of sustainability concerns. For the remaining salt-affected area, this study finds an average biomass yield of 3.1 t dry matter ha^{-1} y^{-1} and a global technical potential of 56 EJ y^{-1} (equivalent to 11% of current global primary energy consumption). If agricultural land is also considered unavailable because of sustainability concerns, the technical potential decreases to 44 EJ y^{-1}. The global economic potential of biosaline forestry at production costs of 2 € GJ^{-1} or less is calculated to be 21 EJ y^{-1} when including agricultural land and 12 EJ y^{-1} when excluding agricultural land. At production costs of up to 5 € GJ^{-1}, the global economic potential increases to 53 EJ y^{-1} when including agricultural land and to 39 EJ y^{-1} when excluding agricultural land. Biosaline forestry may contribute even more significantly to energy consumption in certain regions, *e.g.* Africa. Biosaline forestry has numerous additional benefits such as the potential to improve soil conditions, generate income from previously low-productive or unproductive land, and soil carbon sequestration. These are important additional reasons for investigating and investing in biosaline forestry.

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5.1 INTRODUCTION

In recent years the sustainability of the production and use of energy from plant biomass (bioenergy) has become an issue of global concern (Gallagher, 2008). Key issues are the direct and indirect effects on biodiversity and on food security as well as the greenhouse gas emissions. The use of degraded or low-productive land for the production of bioenergy is often proposed as a solution to these problems. The use of degraded land, which is largely unsuitable for crop production, can reduce (in)direct competition with food production for higher quality land (Gallagher, 2008). The use of degraded land can also increase biodiversity, especially if monoculture and large fields are avoided (CBD, 2008), and improve the greenhouse gas balance by increasing the soil organic matter content as a result of above- and belowground biomass growth (Lal, 2004b; Gibbs *et al.*, 2008). Moreover, the use of previously low-productive areas can contribute to economic growth and create new employment opportunities. However, the use of degraded and low-productive land also has drawbacks that limit its economic attractiveness. Most important are lower yields and higher levels of agricultural inputs such as fertilizers, chemicals, etc., compared to high quality soils.

While previous studies have analyzed bioenergy production from low-productive or degraded land, these studies did not account for either the type and severity of degradation or the impact of degradation on crop yields (Hall *et al.*, 1993; Hoogwijk *et al.*, 2003; van Vuuren *et al.*, 2009; Dornburg *et al.*, 2010). However, these factors can be crucial for the proper design of energy crop production systems and the performance of these systems. In addition, limited attention has been paid to the present use and vegetation cover of degraded and low-productive land. A more in-depth analysis of biomass production in relation to the type and degree of land degradation and to current use of degraded land would allow a better estimation of the potentials. Nijsen *et al.* (submitted) made a first attempt at such an analysis for human-induced degradation and found that the potential of woody crops on degraded land not used as forest, cropland, or pastoral land amounts to 30 to 40 EJ y^{-1}. However, Nijsen *et al.* (submitted) do not account for salt-affected soils nor for any natural degradation although human-induced salt-affected soils are estimated to amount to 76 Mha (Oldeman *et al.*, 1991) while natural and human-induced salt-affected soils combined are estimated between 400 Mha to 960 Mha (van Oosten and de Wilt, 2000; Wood *et al.*, 2000; FAO, 2001b; FAO, 2008b), depending on the datasets and the classification systems used. In addition, salinization of agricultural land continues to occur mainly as a result of mismanagement of irrigated soils, and the annual rate of new irrigation-induced salinization is estimated at 0.25 to 0.5 Mha globally (FAO, 2000). Furthermore, salt-affected land poses challenges to conventional agriculture because most agricultural crops are salt-intolerant. Increased salt concentrations impede plant growth by increasing osmotic pressure of the soil solution, which in turn hampers water extraction by plant roots and thereby growth rates (the osmotic effect) and by increasing concentrations of chloride and sodium ions in the plant, which lead to toxicities in the plants and thereby to cell injury and growth reduction (the specific ion effect) (Munns, 2004; Lantzke *et al.*, 2007). Many tree species are less susceptible to soil salinity and sodicity than agricultural crops, and forestry plantations with these species may thus allow the cultivation of salt-affected land (hereinafter, biosaline forestry) that would otherwise not be used or would have low productivity levels. Examples of such salt-tolerant tree species are *Acacia nilotica*, *Casuarina equisetifolia*, *Prosopis juliflora,* and *Tamarix aphylla* (US National Research Council, 1990). Wood from salt-affected soils can be used for nearly any application of wood without modifications, although co-firing it with coal to produce electricity or gasifying it for liquid fuel production is limited due to higher salt content in the wood leading to corrosion of the equipment (Hoek, 2004). Two examples are the use of saline land for the production of biomass for the local pulp and paper industry in the Yellow River Delta region in China (UNDP, 2007) and the use of sodic soils for fuelwood and charcoal production in the northern Indian state of Haryana (Singh, 2008).

Given the large global extent of salt-affected soils, the continued salinization of agricultural land, and the difficulties of using these lands for agricultural production, the present study focuses on the potential of bioenergy production from biosaline forestry. The objective of this study is to estimate the current global technical and economic potential of woody energy crops cultivated on salt-affected land. This is done by first classifying and mapping the different types of salt-affected land and assessing its current use by applying land use/cover data. Next, a tree growth model is constructed to estimate the yields of different salt-tolerant tree species in salt-affected environments. The results of the first and second step are then combined to estimate the technical bioenergy potentials from salt affected land. Finally, the costs of biomass production are calculated and cost-supply curves constructed to evaluate the economic potential of energy crop production on salt-affected soils.

The remainder of this paper is organized as follows. In Section 5.2, the methodology used in the four abovementioned steps is explained. Section 5.3 describes the spatial datasets, the tree requirements used for determining the yields, the cost data used in the economic potential analysis, and all other input data. The results, including the extent and location of salt-affected soils, the yields, and the technical and economic potential of biomass production from salt-affected soils, are presented in Section 5.4. Section 5.5 discusses methodological choices, uncertainties in the data and the results. Section 5.6 concludes the study with final remarks.

5.2 METHODOLOGY

A spatial resolution of 1 arcminute is applied throughout the analyses. All datasets are converted to this resolution.

5.2.1 The extent and location of salt-affected areas

Salt-affected soils are commonly considered to comprise saline, sodic, and saline-sodic soils (US Salinity Laboratory, 1954). *Saline soils* are characterized by the presence of soluble salts in such quantities that they interfere with plant growth

(Ghassemi *et al.*, 1995). They have a high electrical conductivity of the saturated soil extract (ECe) but a low exchangeable sodium percentage (ESP). *Sodic soils* refer to an excessive amount of sodium on the exchange complex of the soil (high ESP), while the total amount of salts is low (low ECe) (Lamond and Whitney, 1992; Ghassemi *et al.*, 1995). Sodic soils often have a high pH (above 8.5). *Saline-sodic soils* contain excessive amounts of soluble salts (high ECe) and have enough exchangeable sodium to affect plant growth (high ESP), while the pH is generally below 8.5 (Lamond and Whitney, 1992).

In this study, the severity levels of saline and sodic soils are based on the existing classification system of the US Salinity Laboratory (1954) and defined based on ECe and ESP, respectively (Table 5.1). Severity levels of saline-sodic soils are defined here based on a combination of the severity levels of saline soils and sodic soils (Table 5.1).

Table 5.1: Characterization of different types of salt-affected land and their severity levels (average for 1m soil depth)

Type of salt-affected land	**Indicator**	**Severity level**			
		Slight	Moderate	High	Extreme
Sodic	ESP (%)	15 – 20	20 – 30	30 – 40	> 40
	ECe (dS m^{-1})	< 4	< 4	< 4	< 4
Saline	ECe (dS m^{-1})	2 – 4	4 – 8	8 – 16	> 16
	ESP (%)	< 15	< 15	< 15	< 15
Saline-sodic	ESP (%), ECe (dS m^{-1})	15 – 20, 4 – 8	15 – 20, 8 – 25 20 – 30, 4 – 16 30 – 40, 4 – 8	15 – 20, > 25 20 – 30, 16-25 30 – 40, 8 – 16 40 – 50, 4 – 8	20 – 30, > 25 30 – 40, > 16 40 – 50, > 8 >50, > 4

Based on this classification, the location of salt-affected land is mapped and the global extent is calculated in a Geographic Information System (ESRI ArcGIS 9.3.1) using the Harmonized World Soil Database (HWSD) (FAO *et al.*, 2008). The HWSD includes soil characteristics for topsoils (0-30 cm) and subsoils (30-100 cm). Average soil salinity and sodicity are calculated by applying weighting factors of 60% for topsoils and 40% for subsoils. These factors are based on the distribution of tree roots in the soil (Vashev *et al.*, 2010). The HWSD mapping units are divided in up to nine soil units. If not all soil units are salt-affected, only the extent of the salt-affected soil units is considered by multiplying the mapping unit’s area by the percentage share of the soil unit.

5.2.2 Yields of forestry plantations on salt-affected soils

The yields of biosaline forestry are determined separately for (sub)tropical and temperate regions. For (sub)tropical climates, the yield estimation model for salt-affected environments in (sub)tropical regions of Vashev *et al.* (2010) is used (Section

5.2.2.1). For temperate climates, a similar method based on a modified version of the Crop and Grass Production Model of Leemans and Van den Born (1994) is applied (Section 5.2.2.2).

5.2.2.1 Yields of forestry plantations on salt-affected soils in (sub)tropical climates

The yields of forestry plantations on salt-affected soils in (sub)tropical regions are calculated using a modified version of the yield estimation model of Vashev *et al.* (2010)*,* which is based on Sys *et al.*'s (1991) refined version of the FAO (1976) Framework for Land Evaluation and matches climate, soil, and terrain requirements of salt-tolerant tree species (hereinafter, tree requirements) suitable for (sub)tropical regions with the characteristics of the land under consideration. Vashev *et al.* (2010) derive the tree requirements for tropical, salt-tolerant tree species from 1) literature, 2) regression analyses using a database of measurements from pot trials and case studies of biomass production on salt-affected soils, and 3) expert judgment.

The following (groups of) land characteristics are distinguished with respect to soil and terrain:

- *topography* (slope gradient),
- *wetness* (internal drainage class),
- *physical soil characteristics* (gravel content, drainage class, soil texture class, gypsum, calcium carbonate content),
- *chemical soil characteristics* (cation exchange capacity of the clay fraction, base saturation, total exchangeable bases, organic carbon, pH (H_2O)), and
- degree of *salinity-alkalinity* (electricity conductivity, exchangeable sodium percentage).

Vashev *et al.* (2010) include three additional land characteristics (flooding, soil depth, and depth of groundwater) for which global data are unavailable or insufficient to be able to include them in the global analysis (see Section 5.5 for a discussion). In addition to land characteristics, the following climatic characteristics are taken into account:

- *rainfall* (annual precipitation, length dry season),
- *temperature* (mean maximum temperature of the warmest month, mean minimum temperature of the coldest month, mean annual temperature), and
- *radiation* (fraction of sunshine hours).

Depending on the tree-specific requirements, ratings between 0 (unsuitable) and 100 (very suitable) are defined, indicating the level of limitation for the growth of the tree species under the given climate and land characteristic. A climate index and a soil and terrain index are then calculated based on the theory that the scarcest resource is the limiting factor for plant growth. This is done by selecting the ratings of the most limiting factor within each group of land and climate characteristics and by multiplying them (equation 2) (Sys *et al.*, 1991).

$$I_{[tropS,\,tropC]} = A \times (B/100) \times (C/100) \times (D/100) \;.... \qquad (2)$$

where I_{tropsS} [unitless] – soil and terrain index; I_{tropsC} [unitless] – climate index; A, B, C, D… - rating of the most limiting factor within each group of land characteristics (topography, wetness, physical soil characteristics, chemical soil characteristics and salinity-alkalinity) and climatic characteristics (rainfall, temperature and radiation).

The climate index and soil and terrain index indicate the impact of climate, soil, and terrain separately. To calculate a land index that combines climate, soil, and terrain characteristics, the climate index is first recalculated into a climate rating (R_C, unitless) following Equation 3 (based on Sys *et al.* (1991)).

$$R_C = \begin{cases} I_{tropC} \times 1.60 & \text{when } 0 \le I_{tropC} \le 25.0 \\ I_{tropC} \times 0.94 + 16.67 & \text{when } 25.0 < I_{tropC} \le 92.5 \\ I_{tropC} & \text{when } 92.5 < I_{tropC} \le 100.0 \end{cases} \qquad (3)$$

The climate rating is then multiplied by the soil and terrain index to determine a land index (LI_{trop}, unitless) (Equation 4), which represents the suitability of the land for the given tree species and is relative to the constraint-free yield.

$$LI_{trop} = R_C \times (I_{tropS}/100) \qquad (4)$$

Values for the LI_{trop} range between 0 (not suitable) and 100 (very suitable). To estimate the actual yield (Y_{trop}, t dry matter (dm) ha^{-1} y^{-1}), the LI_{trop} is multiplied with the constraint-free yield (Y_{max}, t dm ha^{-1} y^{-1}):

$$Y_{trop} = Y_{max} \times (LI_{trop}/100) \qquad (5)$$

The constraint-free yield of the (sub)tropical tree species is approximated by applying the maximum yields recorded in the literature. A management factor that accounts for differences in theoretical and actual yields is not applied in the tropical model because the yields used in the study refer either to actual yields obtained at plantations (Acacia nilotica (Maguire *et al.*, 1990) and Prosopis juliflora (Pasiecznik *et al.*, 2001)) or to a calculated potential yield that accounts for the harvest index (Eucalyptus camaldulensis (van den Broek *et al.*, 2001)). Results are generated for three salt-tolerant species, which have shown promising yields in pot trials, field experiments, and literature, and for which sufficient data is available (Vashev *et al.*, 2010). These species are *Eucalyptus camaldulensis*, *Acacia nilotica,* and *Prosopis juliflora*. For the potential analysis, the yield in each grid cell is defined by the species with the highest yield.

5.2.2.2 Yields of forestry plantations on salt-affected soils in temperate climates

The yields of forestry plantations on salt-affected soils in temperate climates (Y_{temp})

are estimated using a modified version of the Crop and Grass Production Model (CGPM) of the Integrated Model to Assess the Global Environment (IMAGE) (Leemans and Born, 1994; MNP, 2006). In the CGPM, climate-constraint yields (Y_{clim}) are calculated and multiplied by a soil reduction factor that accounts for soil and terrain limitations to crop production. This soil reduction factor (hereinafter referred to as soil index, I_{tempS}, in line with the terminology used in the (sub)tropical model) is determined as follows:

$$I_{tempS} = 0.005 \times R_g \times (R_{nr} + R_{sy} + R_{ro} - R_g) \qquad (6)$$

where I_{tempS} [unitless] – soil index; $R_{nr,\ sy,\ ro}$ [unitless] – rating of the most limiting factor within each of the three soil quality indicators: *nutrient retention and availability* (R_{nr}; fertility), *level of salinity, alkalinity and toxicity* (R_{sy}; salinity, pH, sodicity) and *rooting conditions* for the plants (R_{ro}; rooting depth, drainage); and R_g [unitless] – minimum of R_{nr}, R_{sy}, and R_{ro}. All ratings range between 0 (unsuitable) and 100 (very suitable).

In order to better account for the salt-tolerance of some tree species, the first modification applied to the CGPM by the present study is the way the ratings for I_{tempS} are calculated. In the original model, the ratings for each crop type are defined per soil class. In the present study this is done only for the rating of nutrient retention and availability and the rating of rooting depth. The other ratings are defined based on the average tree requirements of the three species used in the (sub)tropical model assuming that the soil and terrain requirements of temperate tree species are similar to those of tropical species. This assumption is made because tree requirements for salt-tolerant, temperate species do not yet exist. However, salt-affectedness is the main parameter in this study, and literature on the salt-tolerance of temperate tree species, including those applied in the IMAGE model (*e.g.* poplar and willow species), indicates that various temperate tree species are also salt-tolerant (Hayward and Bernstein, 1958; Khamzina, 2006; Chen and Polle, 2010).

The biomass yield for salt-affected soils in temperate regions (Y_{temp}) is then calculated by multiplying the climate-constraint yield from the CGPM by the soil index and a management factor (equation 7). A management factor of 0.7 is applied to account for differences in theoretically feasible and actual yields (Hoogwijk *et al.*, 2005).

$$Y_{temp} = Y_{clim} \times (I_{tempS}/100) \times MF \qquad (7)$$

where Y_{clim} [t dm ha^{-1} y^{-1}] – climate constraint yield; I_{tempS} [unitless] – soil index (equation 5); and MF [unitless] – management factor.

A second modification to the CGPM is made with respect to the soil database applied for calculating the soil index. The HWSD (FAO *et al.*, 2008) is used because it is more updated and detailed than the DSMW (FAO, 2003a) used in the original model.

5.2.3 Technical potential of biomass production on salt-affected soils

The technical potential of biomass production on salt-affected soils is determined per grid cell by multiplying the available salt-affected area by the yield corresponding to the climate and soil characteristics of the grid cell. Salt-affected land is assumed to be available if it is not classified as forest, wetland, unsuitable areas (*e.g.* urban areas), or (inter)nationally protected areas. Agricultural land is not excluded in the potential assessment because conversion to a forestry plantation can reduce the risk of further degradation of the land and may even help improve the soil (Singh *et al.*, 1994; Singh, 1995; Bell, 1999; Lambert and Turner, 2000). However, the use of agricultural salt-affected land for biomass production may not be desirable for various reasons, most importantly food insecurity and (in)direct land use change. Therefore, the fraction of the technical potential originating from agricultural land is distinguished.

5.2.4 Economic potential of biomass production on salt-affected soils

The economic potential is in this study defined as the part of the technical potential that can be produced at a certain (attractive) cost level. Due to the large number of biomass applications and conversion technologies, it is not possible to determine the competitiveness for all combinations of applications and conversion technologies. Instead, the focus is on the cost of the biomass production. A figure of 2 € GJ^{-1} or below is assumed to be an attractive range for the costs of biomass feedstock production because at this level large scale production of second generation liquid biofuels is expected to become competitive with conventional gasoline, assuming that technological developments will be stimulated (Hamelinck and Faaij, 2006). Co-firing biomass with coal for electricity production is also competitive at this level given that the current price of coal is 2.3 € GJ^{-1} (US EIA, 2010). A range of 2 to 5 € GJ^{-1} can still be considered attractive for certain applications, but attractiveness depends heavily on the price of oil if the biomass is intended for energy and oleochemical purposes. More detailed and site-specific analysis will be required on whether the applications of biomass from salt-affected soils are indeed economically feasible.

The economic potential is determined by constructing cost-supply curves for biomass production from biosaline forestry. These curves are made by ranking the geographic potential as a function of production costs per grid cell. The farm-gate production costs (in US$ GJ^{-1}) are calculated by applying discounted values for costs and biomass yields because costs and benefits from biomass production are distributed unequally over time (van den Broek *et al.*, 2000; Smeets *et al.*, 2009c). Converting physical units (*i.e.* the yield) into annuities may be uncommon, but the concept is essentially the same as converting costs into annuities because physical units also represent monetary values. The production costs are determined as follows:

$$P_{\text{cost}} = \sum_{t=0}^{n} \frac{(C_t)}{(1+r)^t} \times EC^{-1} \times \left(\sum_{t=0}^{n} \frac{X_t}{(1+r)^t} \right)^{-1} \tag{8}$$

where P_{cost} [€ GJ^{-1}] - costs of production, C_t [€ ha^{-1}] - costs of the forestry plantation in year t, X_t [t ha^{-1}] - yield of wood in year t, EC [GJ t^{-1}]- energy content of woody biomass, r [%] - discount rate, and n [y] - lifetime of the project.

The range of forestry systems suitable for salt-affected soils varies with respect to factors such as the management system (fertilizer application rate, use of irrigation, level of mechanization), the tree species, the use of intercropping, and the planting density. The economic attractiveness of each system depends primarily on the price of biomass, land, labour, capital and other inputs; the availability of infrastructure; and the costs of transportation. A detailed evaluation to determine the optimal systems in each grid cell is not possible on a global scale due to a lack of data. Instead, a generalized forestry system that includes all elements and cost items of a typical forestry plantation is defined. The generalized production system assumes two rotation periods of ten years each. The establishment phase involves soil preparation, planting of trees (at a tree density of approximately 800 trees per hectare), weeding, pruning, and fertilizing. Irrigation is considered only during the establishment phase to improve tree survival and not as part of the maintenance of the plantation.

The maintenance of forestry plantations requires wedding, fertilizing, and pruning. Weeding is assumed to be required only in the first three years after establishment and in the first year after the harvest. The nitrogen, phosphorous, and potassium fertilizer requirements are determined by means of a nutrient balance methodology, which assumes that the nutrients taken up by the crop during its growth must be replenished by fertilizers in order to maintain the soil's nutrient composition (De Wit and Faaij, 2010). While this is a simplification of the actual practice, it enables a fair comparison of fertilizer requirements in different regions with different productivities.

Harvesting and in-field transportation can be a manual labour-based system (using only chainsaws and manpower), a fully mechanized system (using large, self propelled harvesters, forwarders, and tractors), or one of various intermediate systems. The choice of the system depends primarily on the price of labour and machinery. Because the type of system applied affects the costs of harvesting and transportation of the biomass to the edge of the field, this study defines three harvest systems, namely, one manual, one fully mechanized, and one intermediate system, to account for the many different possible levels of mechanization. In this study, the definition of the three systems is based on data on labour input and machinery costs from the literature (see Table 5.5). A constant price of capital is assumed across all countries meaning that the price of agricultural labour determines which harvesting system is used in each country.

Another important factor in the production cost of biomass is land rent. The rent of degraded land depends on many factors such as the severity of the degradation, the distance to cities, and available infrastructure. Because only few data points are available, regional costs of land rent are taken from Hoogwijk *et al.* (2009) and corrected for the lower value of salt-affected land compared to high quality agricultural land. The correction factor is based on the ratio of average yields of salt-affected soils and average forestry plantation yields in the global potential study of Hoogwijk *et al.* (2009) Although this is a rough approach, it provides an initial estimate that can be used in this study.

5.3 INPUT DATA

Although the scope of this assessment is global, results for 17 world regions (IMAGE team, 2001) are generated in order to show the impact of regional differences in soil and climate (and thereby in yield) and in the price of land, labour, and inputs. Regional or country specific data are included whenever available.

5.3.1 Spatial datasets

The extent and location of salt-affected soils worldwide are determined with the HWSD (FAO *et al.*, 2008). Current land use/cover of salt-affected land is assessed by applying the Global Land Cover 2000 database (GLC2000) (2003). Nationally and internationally protected areas are accounted for by the World Database on Protected Areas (UNEP-WCMC and IUCN WCPA, 2009).

All soil parameters used in the yield model are extracted from the HWSD (FAO *et al.*, 2008). Slopes are mapped with the median slope gradient map of IIASA and FAO (2000). All climate parameters, except the *length of dry season*, are extracted from the CRU TS 2.1 dataset (Mitchell and Jones, 2005), applying the average between 1981 and 2002. The parameter *length of dry season* is determined using monthly precipitation data from the CRU TS 2.1 dataset (Mitchell and Jones, 2005) and monthly reference evapotranspiration from FAO (FAO GIS Unit, 2000). (Sub)tropical and temperate regions are distinguished using the Thermal Climate Zones Map from FAO (FAO GIS Unit, 2007).

5.3.2 Yields

The tree requirements applied in determining the soil and terrain index and the climate index are presented in Table 5.2 and Table 5.3, respectively. Constraint-free yields for the harvested biomass of the (sub)tropical tree species (Y_{max}) are 41 t dm ha^{-1} y^{-1} for *Acacia nilotica* (Maguire *et al.*, 1990), 38 t dm ha^{-1} y^{-1} for *Eucalyptus camaldulensis* (van den Broek *et al.*, 2001) and 39 t dm ha^{-1} y^{-1} for *Prosopis juliflora* (Pasiecznik *et al.*, 2001) In the analyses an average lower heating value of woody biomass of 18.4 GJ t^{-1} dm is assumed for all species (De Wit and Faaij, 2010).

Table 5.2: Climate requirements for Eucalyptus camaldulensis, Acacia nilotica and Prosopis juliflora (Vashev *et al.*, 2010)

		Rating					
	Species [a]	100	90	72.5	50	32.5	0
Rainfall [b]							
Annual Precipitation (mm)	*A. nilotica*	≥ 1200	1200 - 1000	750 - 1000	500 - 750	200 - 500	0 - 200
Annual Precipitation (mm)	*E. camald.*	≥ 2500	1000 - 2500	600 - 1000	400 - 600	250 - 400	0 - 250
Annual Precipitation (mm)	*P. juliflora*	≥ 1200	750 - 1200	550 - 750	300 - 550	100 - 300	0 - 100
Rating		100	90	72.5	50	32.5	12.5
Length dry season (months) [c]	*A. nilotica*	0 - 6	6 - 7	7 - 8	8 - 9	9 - 10	10 - 12
Length dry season (months) [c]	*E. camald.*	0 - 1	1 - 2	2 - 4	4 - 7	7 - 8	8 - 12
Length dry season (months) [c]	*P. juliflora*	0 - 6	6 - 7	7 - 8	8 - 10	10 - 11	11 - 12
Temperature							
Mean max temp. (°C)	*A. nilotica*	25 - 28	28 - 39	39 - 47	47 - 50	50 - 55	> 55
Mean max temp. (°C)	*E. camald.*	22 - 30	30 - 35	35 - 41	41 - 44	44 - 47.5	> 47.5
Mean max temp. (°C)	*P. juliflora*	20 - 30	30 - 34	34 - 42	42 - 50	50 - 55	> 55
Mean annual temp. (°C)	*A. nilotica*	24 - 28	19 - 24 28 - 34	17 - 19 34 - 39	15 - 17 39 - 45	13 - 15 45 – 50	< 13 > 50
Mean annual temp. (°C)	*E. camald.*	20 - 24	24 - 26 18 - 20	26 - 29 15 - 18	29 - 32 12 - 15	32 - 38 7-12	> 38 < 7
Mean annual temp. (°C)	*P. juliflora*	20 - 30	30 - 35 18 - 20	35 - 38 16 - 18	38 - 42 14 - 16	42 - 45 12 – 14	> 45 < 12
Mean min. temp (°C)	*A. nilotica*	19 - 25	25 - 34 15 - 19	10 - 15	6 - 10	4 - 6	< 4
Mean min. temp (°C)	*E. camald.*	18 - 24	24 - 28 14 - 18	10 - 14	7 - 10	1 - 7	< 1
Mean min. temp (°C)	*P. juliflora*	20 - 25	16 - 20 25 - 35	12 - 16	8 - 12	5 - 8	< 5
Radiation							
Fraction of sunshine hours (-)	All species	0.7 - 1.0	0.5 - 0.7	0.0 - 0.5			

a – A. nilotica - Acacia nilotica, E. camald. - Eucalyptus camaldulensis, P. juliflora - Proposes juliflora, All species – Acacia nilotica, Eucalyptus camaldulensis, Proposes juliflora;

b – The annual precipitation rating is not taken into account by Vashev *et al.* (2010) because their study assumes that all water requirements are met by groundwater. This was done because salt-affected land is often located in arid and semi-arid regions where tree growth relies mainly on groundwater. However, as global groundwater datasets are not available, the present study assumes that water requirements are met by precipitation only. Therefore, in areas where groundwater tables are close to the surface, the potentials are underestimated.

c – The length of dry season (in months) is determined by comparing monthly precipitation (P) with monthly potential evapotranspiration (PET). When P is less than half of PET, the month is considered as part of the dry season.

Table 5.3: Soil and terrain requirements for Eucalyptus camaldulensis, Acacia nilotica and Prosopis juliflora (Vashev *et al.*, 2010)

		Rating					
	Species[a]	100	90	72.5	50	32.5	12.5
Topography							
Slope (%)	All species	0 - 4	4 - 8	8 - 16	16 - 30	30 – 50	50 - 100
Wetness							
Drainage class (-) [b]	*A. nilotica* *P. juliflora*	E, S, W, M		I	P	V	
Drainage class (-) [b]	*E. camald.*	E, S, W, M, I	P	V			
Physical soil characteristics							
Gravel content (volume %)	All species	0 - 3	3 - 15	15 - 35	35 - 55		55 - 100
$CaCO_3$ (%)	*A. nilotica* *P. juliflora*	0 - 20	20 - 30	30 - 40	40 - 60	60 - 100	
$CaCO_3$ (%)	*E. camald.*	0 - 6	6 - 15	15 - 25	25 - 35	35 - 100	
Gypsum (%)	All species	0 - 3	3 - 5	5 - 10	10 - 20		20 - 100
Texture class (-) [c]	*A. nilotica*	6, 7, 9, 10, 11	4, 5, 8, 12	1, 2, 3, 13			
Texture class (-) [c]	*E. camald.*	4, 6, 7, 9, 10, 11	5, 8, 12	2	3	1, 13	
Texture class (-) [c]	*P. juliflora*	4, 5, 6, 7, 8, 9, 10, 11, 12	2, 3	1, 13			
Chemical soil characteristics							
Cation exchange capacity of clay fraction (cmol kg^{-1} clay)	All species	≥ 24	16 - 24		< 16		
Base saturation (%)	All species	50 - 100	35 - 50	20 - 35	0 - 20		
Total exchangeable bases (cmol kg^{-1} soil)	All species	≥ 4.0	2.8 - 4.0	1.6 - 2.8	0.0 - 1.6		

Table 5.3: Continued

Organic carbon (%)	*E. camald.* *A. nilotica*	≥ 1	0.4 - 1.0	0.1 - 0.4	0.01 - 0.1	0 - 0.01	
Organic carbon (%)	*P. juliflora*	≥ 1	0.2 - 1.0	0.1 - 0.2	0.01 - 0.1	0 - 0.01	
pH H_2O	*A. nilotica*	7.0 - 7.5	5.5 - 7.0	4.0 - 5.5	3.0 - 4.0	9.4 - 9.8	< 3.0
			7.5 - 8.5	8.5 - 9.0	9.0 - 9.4		> 9.8
pH H_2O	*E. camald.*	6.5 - 7.5	5.5 - 6.5	4.5 - 5.5	3.5 - 4.5	3.0 - 3.5	< 3.0
			7.5 - 8.7	8.7 - 9.0	9.0 - 9.2	9.2 - 9.4	> 9.4
pH H_2O	*P. juliflora*	6.7 – 8.1	5.5 - 6.7	4.0 - 5.5	3.0 - 4.0	9.5 - 10.2	< 3.0
			8.1 - 8.7	8.1 - 9.2	9.2 - 9.5		> 10.2
Degree of salinity - alkalinity							
ECe (dS m^{-1})	*A. nilotica*	0.0 - 3.8	3.8 - 7.2	5.2 - 11.3	11.3 - 16.1	16.1 - 19.8	> 19.8
ECe (dS m^{-1})	*E. camald.*	0.0 - 2.0	2.0 - 8.0	8.0 - 12.0	12.0 - 14.0	14.0 - 16.0	> 16.0
ECe (dS m^{-1})	*P. juliflora*	0.0 - 3.0	3.0 - 6.1	6.1 - 14.0	14.0 - 20.3	20.3 - 25.0	> 25.0
ESP (%) [d]	*A. nilotica*	0 - 60	> 60				
ESP (%) [d]	*E. camald.*	0 - 50	> 50				
ESP (%) [d]	*P. juliflora*	0 - 70	> 70				

a – A. nilotica - Acacia nilotica, E. camald. - Eucalyptus camaldulensis, P. juliflora - Proposes juliflora, All species – Acacia nilotica, Eucalyptus camaldulensis, Proposes juliflora;

b – Drainage classes (Soil Survey Division Staff, 1993): E – excessively drained, S – somewhat excessively drained, W – well drained, M – moderately well drained, I – imperfectly drained, P – poorly drained, V – very poorly drained;

c – Texture class (Soil Survey Division Staff, 1993): 1 – clay (heavy), 2 – silty clay, 3 – clay, 4 – silty clay loam, 5 – clay loam, 6 – silt, 7 – silt loam, 8 – sandy clay, 9 – loam, 10 – sandy clay loam, 11 – sandy loam, 12 – loamy sand, 13 – sand;

d – Nearly all semi-arid and arid soils with high ESP (sodic soils) also have a high pH value (Abrol *et al.*, 1988). If stringent tree requirements for both pH and ESP are applied, the effect of pH and ESP on the yield would be double counted. As a result, the ESP requirements are made less restrictive.

5.3.3 Production costs

The costs of establishment of forestry plantations in different world regions are taken from Strengers *et al.* (2008) and vary between 320 and 506 € ha^{-1} (Table 5.4). The costs of land rent are based on Hoogwijk *et al.* (2009) but corrected by the ratio of average yields on salt-affected soils (as determined in the present study to be 3.1 t dm ha^{-1} yr^{-1}) and average forestry plantation yields (as determined in the global potential study of Hoogwijk *et al.* (2009) to be 7.5 t dm ha^{-1} yr^{-1}). Regional salt-affected land rent is presented in Table 5.4.

Table 5.4: Land rent, establishment and maintenance costs, per world region [a]

	Land rent (Hoogwijk *et al.*, 2009) € ha^{-1} y^{-1}	**Establishment costs** (Strengers *et al.*, 2008) € ha^{-1}	**Maintenance** (Riegelhaupt, 2001; Guitart and Rodriguez, 2010; Lopez *et al.*, 2010) € ha^{-1} y^{-1}
Canada	24	426	31
USA	56	441	33
C America	46	506	37
S America	44	369	27
N Africa	10	426	31
W Africa	8	426	31
E Africa	7	320	24
S Africa	29	426	31
W Europe	47	329	24
E Europe	25	329	24
F USSR	10	363	27
M East	11	490	36
S Asia	47	490	36
E Asia	104	467	34
SE Asia	55	481	36
Oceania	5	369	27
Japan	247	326	24

a - Definition of world regions is based on the IMAGE team (2001).

An average cost of maintenance (excluding the cost of fertilizers) is estimated to be 30 € ha^{-1} y^{-1} based on studies by Riegelhaupt (2001), Lopez *et al.* (2010), and Guitart and Rodriguez (2010). Regional differences in maintenance costs are assumed to be similar to the regional differences in establishment costs. Thus, regional maintenance costs are determined by multiplying the average maintenance cost with the ratio of regional establishment cost to average establishment costs (Table 5.4). The fertilizer costs are calculated by assuming that the nutrients in the harvested biomass need to be replaced, whereby a nutrient content of 4.40 kg N (t dm)$^{-1}$, 0.45 kg P (t dm)$^{-1}$ and 2.70 kg K (t dm)$^{-1}$ of wood, fertilizer factors of 1 kg N kg^{-1} N, 2.3 kg P_2O_5 kg^{-1} P and 1.2 kg

K_2O kg^{-1} K (De Wit and Faaij, 2010), and fertilizer costs from FAOSTAT (2009) are applied. To determine the harvesting costs, labour requirements and machinery costs of the three harvest systems are defined as shown in Table 5.5. Country-specific data on the price of labour are taken from LabourSTA (ILO, 2009). A minimum price of labour of 0.25 € h^{-1} is assumed.

Table 5.5: Labour input and machinery costs for harvest systems with different levels of mechanization (based on WSRG, 1994; Perlack *et al.*, 1997; van den Broek *et al.*, 2000; FAO, 2008a; Smeets *et al.*, 2009c; Smeets and Faaij, 2010)

Level of mechanization	**Labour input** h (t dm)$^{-1}$	**Machinery costs** € (t dm)$^{-1}$
Manual	15.0	0.7
Intermediate	8.6	3.9
Fully mechanized	0.5	32.7

Table 5.6: The extent of salt-affected soils, by type and severity of salt-affectedness

Severity level	**Unit**	**Type**			**Total** [a]	**Share (%)**
		Saline	Sodic	Saline-sodic		
Slight	1000 ha	606	124	6	735	65
Moderate	1000 ha	69	147	11	228	20
High	1000 ha	4	13	36	52	5
Extreme	1000 ha	4	5	105	113	10
Total [a]	1000 ha	683	288	157	1128	
Share	%	60	26	14		

a - Rows and columns may not actually sum to the given total due to rounding.

5.4 RESULTS

5.4.1 Extent and location of salt-affected areas

The global extent of salt-affected land, as calculated from the HWSD, amounts to 1,128 Mha (Table 5.6). This is slightly higher than previous estimates. For example, Szabolcs (1989) estimates salt-affected land to be 955 Mha and FAO (2008b), 831 Mha. Insufficient information is available to determine the exact reasons for these discrepancies but such reasons could include different definitions of salt-affectedness and the application of different soil datasets. Global salt-affected soils are mainly saline, amounting to 60% of all salt-affected soils (Table 5.6). Sodic soils account for 26% and saline-sodic soils for 14%. The majority of salt-affected soils is slightly affected (65%), followed by 20% moderately, 10% extremely, and 5% highly salt-affected soils.

The mapping of salt-affected land shows that in nearly all world regions salt-affected soils are found, although the extent and severity vary among regions (Figure

5.1, Table 5.7). Regions with the largest salt-affected land areas are the Middle East (189 Mha), Australia (169 Mha), North Africa (144 Mha), and the former USSR (126 Mha) (Table 5.7). Excluding forests, wetlands, unsuitable areas, and (inter)nationally protected areas results in 971 Mha (or 86% of the total extent of salt-affected land) available for consideration in the analysis of the potentials (Table 5.7).

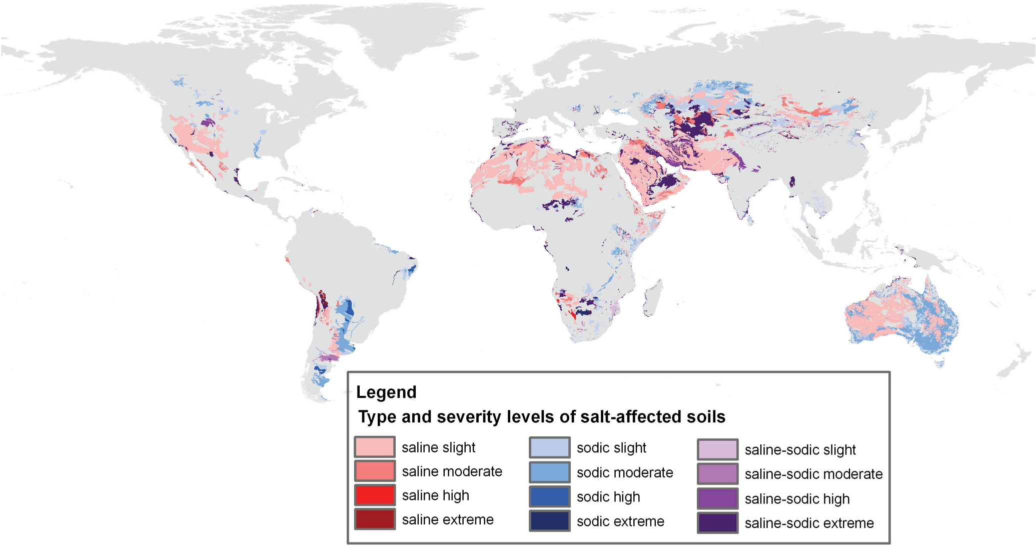

Figure 5.1: Global salt-affected soils, by type and severity (based on data from the HWSD (FAO *et al.*, 2008)) [a]

a - This map indicates the location of salt-affected soils worldwide but does not properly represent their areal extent as a result of multiple soil units per mapping unit of the HWSD. Multiple soil units are defined because mapping units are not generally homogeneous in soil characteristics. Up to nine soil units may be defined per mapping unit, and the map depicts the whole mapping unit to be salt-affected even if only one of the soil units is salt-affected. For the areal extent of salt-affected soils see Table 5.6.

5.4.2 The global technical biomass production potential from salt-affected areas

Biomass yields on salt-affected soils (Figure 5.2) range between 0 and 27 t dm ha^{-1} y^{-1} with the average yield being 3.1 t dm ha^{-1} y^{-1}. Yield differences are explained primarily by the severity of salt-affectedness (see Figure 5.1), but climate, particularly precipitation, is obviously an important factor as well.

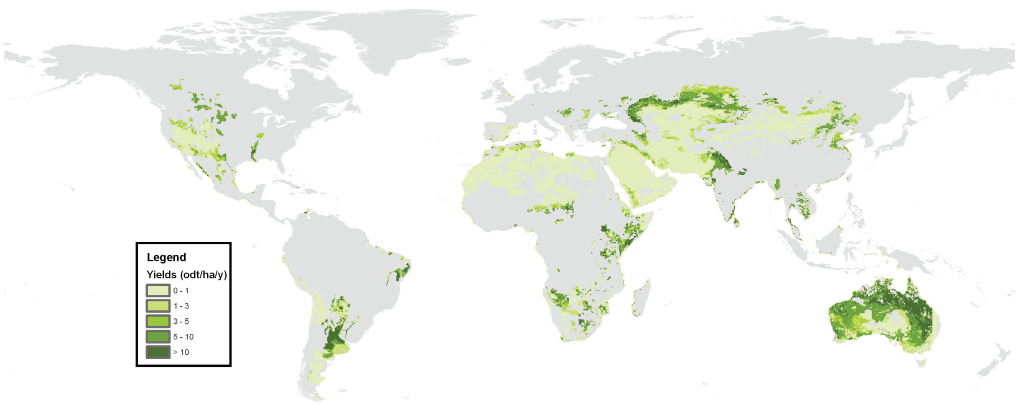

Figure 5.2: Projected yields on salt-affected soils

The total global technical biomass production potential of biosaline forestry is calculated to be 56 EJ y^{-1} (3,114 million t dm y^{-1}) (Table 5.7), which represents approximately 11% of the current global primary energy use of approximately 514 EJ y^{-1} (OECD/IEA, 2010c). The regional breakdown of the technical potential shows that Oceania has the highest potential with 20 EJ y^{-1}, which is followed by the former USSR region with 10 EJ y^{-1}, South America with 5 EJ y^{-1}, and East Africa with 5 EJ y^{-1} (Table 5.7). The high potential in Australia is primarily due to the very large amount of land that is salt-affected (169 Mha, Table 5.7), most of which is only slightly salt-affected. The low severity explains an average yield (7.6 t ha^{-1} y^{-1}) that is more than twice the global average yield.

The breakdown of the potential by severity level and land use/cover class indicates that the largest potentials can be found on slightly and moderately affected areas that are currently covered by shrubs and herbaceous vegetation (66%) (Table 5.8). Of this potential, 26% comes from agricultural land, which takes place primarily on slightly and moderately affected soils. Thus, if current agricultural land is considered unavailable for biomass production because of sustainability concerns, the technical potential decreases to 42 EJ y^{-1}. Highly and extremely salt-affected soils combined account for only 6% (or 4 EJ y^{-1}) of the technical potential (Table 5.8). The technical potential broken down by land use/cover classes, severity levels, and the 17 world regions is presented in the appendix (Table 5.9).

Table 5.7: The extent of salt-affected soils and the technical and economic biomass production potential, by region

Region	**Salt-affected land**	**Salt-affected land excl. forest, wetlands, unsuitable, high biodiversity areas**	**Technical potential** [b]	**Economic potential**	
				≤ 2 € GJ^{-1}	≤ 5 € GJ^{-1}
	Mha	Mha	EJ y^{-1}	EJ y^{-1}	EJ y^{-1}
Canada	7	5	0.7	0.0	0.7
USA	77	58	2.9	0.0	2.0
Central America	5	4	0.3	0.0	0.2
South America	84	57	5.4	3.7	4.9
North Africa	161	157	1.1	0.6	1.1
West Africa	83	76	0.8	0.7	0.8
East Africa	56	43	5.1	5.0	5.1
South Africa	22	19	2.0	1.2	1.9
West Europe	1	1	0.0	0.0	0.0
East Europe	2	1	0.2	0.0	0.2
Former USSR	126	117	10.0	6.3	9.7
Middle East	176	158	1.8	0.6	1.5
South Asia	52	45	2.8	2.4	2.7
East Asia	98	83	2.6	0.0	2.1
Southeast Asia	6	5	0.5	0.2	0.5
Oceania	169	144	20.2	0.0	19.7
Japan	0	0	0.0	0.0	0.0
World[a]	1,126	971	56.2	20.8	52.8

a - Columns may not actually sum to the given total due to rounding.

b - The technical and economic potential refers to salt-affected land not classified as forests, wetlands, unsuitable areas, or (inter)nationally protected areas.

Table 5.8: The technical biomass production potential, by severity level and land use/cover class

Land use/cover	**Severity level**				**Total** [a]	**Share**
	slight	moderate	high	extreme		
	EJ y^{-1}	EJ y^{-1}	EJ y^{-1}	EJ y^{-1}	EJ y^{-1}	%
Agriculture	7.9	4.8	1.3	0.6	14.6	26
Bare areas	2.2	0.7	0.1	0.4	3.4	6
Shrub and herbaceous cover	26.8	10.2	0.5	0.8	38.3	68
Total [a]	36.8	15.7	1.9	1.8	56.2	
Share (%)	65	28	3	3		

a - Rows and columns may not actually sum to the given total due to rounding.

5.4.3 Global economic biomass production potential from salt-affected areas

The average production cost of tree biomass from salt-affected soils is 4.0 € GJ^{-1}, but large regional and intraregional differences in production costs exist (Figure 5.3).

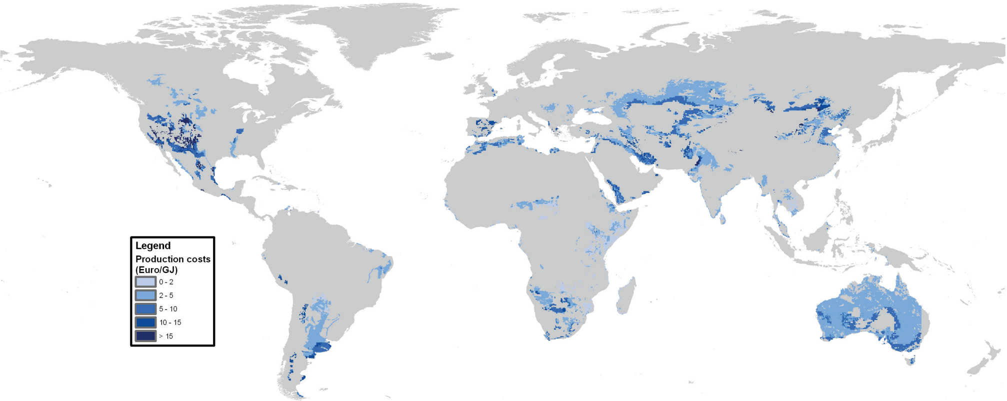

Figure 5.3: Production costs of woody biomass from salt-affected soils

The global economic potential analysis for biomass production on salt-affected soils indicates that there is an economic potential of biomass production from salt-affected soils (when including agricultural land) of 21 EJ y^{-1} (or 4% of global primary energy consumption) at production costs of 2 € GJ^{-1} or less (Table 5.7). The economic potential increases significantly, to 53 EJ y^{-1}, when biomass produced at costs of 5 € GJ^{-1} or less are included. If agricultural land is excluded, the economic potential of biosaline forestry decreases to 12 EJ y^{-1} at production costs of 2 € GJ^{-1} or less and to 39 EJ y^{-1} at production costs of 5 € GJ^{-1} or less.

Global cost supply curves by severity of affectedness (Figure 5.4(a)) confirm that the largest share of the potential comes from the least salt-affected soils. Of the 21 EJ y^{-1} at production costs of 2 € GJ^{-1} or less, 19 EJ y^{-1} (88%) are from slightly and moderately salt-affected soils while only 2 EJ y^{-1} are from highly or extremely salt-affected soils. This trend is even more extreme for the economic potential at production costs of 5 € GJ^{-1} or less, where slightly and moderately affected soils account for 93% of the potential. The global cost supply curves by land use/cover class (Figure 5.4(b)) indicate that biomass production on salt-affected land with shrub and herbaceous cover has the highest economic potential at production costs up to both 2 € GJ^{-1} and 5 € GJ^{-1}.

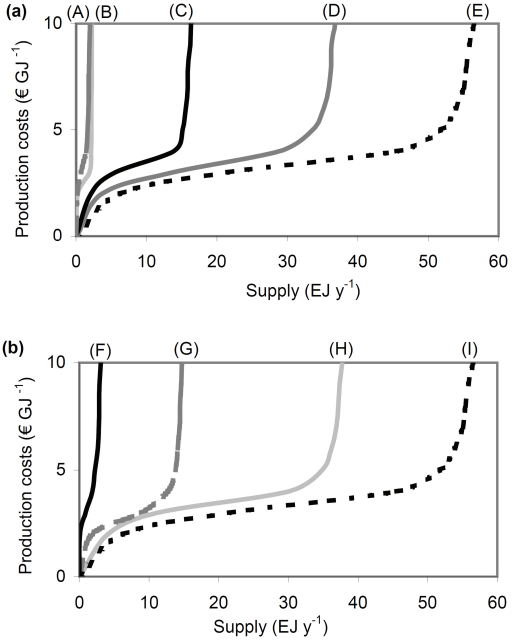

Figure 5.4: Global cost-supply curves for salt-affected soils, by (a) severity and (b) land use/cover

(a) Severity levels: (A) – extreme; (B) – high; (C) – moderate; (D) – slight; (E) – total

(b) Land use/cover classes: (F) – bare areas; (G) – agriculture; (H) shrub and herbaceous cover; (I) – total

5.5 DISCUSSION

The Harmonized World Soil Database (HWSD) (FAO *et al.*, 2008) provides the basis for the mapping of the location of salt-affected land and calculation of global extent. Although the HWSD is the most comprehensive, detailed, and updated global soil database currently available, it has shortcomings associated with compiling a global dataset from a range of sources and to an uneven geographical distribution of soil profile analyses (for a discussion of the shortcomings see FAO, IIASA, ISRIC, ISS-CSA and JRC (2008)). In addition, the HWSD is less reliable for regions that are presented by the Digital Soil Map of the World (DSMW) such as North America, Australia, West Africa, and South Asia. For both North America and Australia, updates are likely forthcoming given that newly updated datasets are now available for these regions (FAO *et al.*, 2008). The case of South Asia, particularly India and Pakistan, indicates the importance of such updates. For example, calculating the extent of salt-affected land based on the HWSD suggests 11 Mha in India and 9 Mha in Pakistan, but Vashev *et al.* (2010) find the extent of salt-affected land to be only 7 Mha in India and 3 Mha in Pakistan (although the findings for Bangladesh (1 Mha) are similar to the HWSD data).

Therefore, future updates to the HWSD must be made, its effect on the global potential bioenergy production on salt-affected soils must be evaluated, and regional assessments must be conducted.

The availability of salt-affected land for biosaline forestry is determined in this study by its current land use/cover and the extent of areas of high biodiversity. Agricultural land (both for crop production and livestock grazing) is not excluded in the potential analyses because conversion to a forestry plantation may prevent further salinization/sodification of the land and may even provide soil improvements (Singh *et al.*, 1994; Singh, 1995; Bell, 1999; Lambert and Turner, 2000). This study found that agricultural land accounts for 13% of the technical potential, but the effect of agriculture on land availability may be even larger considering that extensive agricultural land use such as livestock grazing commonly takes place on land with shrub and herbaceous cover. This is not yet accounted for in this study because seasonal and inter-annual variability in grazing and the low quality of census data on pastureland makes livestock grazing difficult to demarcate (Ramankutty *et al.*, 2008). Future assessments of the actual availability of salt-affected soils for biomass production should account for grazing and determine the effect that grazing would have on the availability of salt-affected land and technical biomass potential. It should be noted that biomass production on salt-affected soils can be combined with food and feed/forage production by, for example, intercropping, rotational woodlots, and hedge rows. The potential of such combined systems should also be assessed given that they may be more preferable with respect to ensuring food security. In addition to current land use/cover, it is also important to account for future developments in land use and the impact on the extent and availability of salt-affected land for biosaline forestry. An important factor in future land use and land use change is likely to be the increasing demand for land for agricultural production to meet the growing world population's demand for food and dietary changes. As highly productive land becomes scarcer, agriculture may have to increasingly rely on low productive and degraded (including salt-affected) land and may reduce the availability for biosaline forestry. In addition to current land use/cover as an indicator of the (un)availability of salt-affected land, salt-affected land may also be considered unavailable as a result of high biodiversity. This study accounted for high biodiversity areas by excluding nationally and internationally protected areas. However, little is known about the actual biodiversity levels of salt-affected land. Future research should assess this aspect and its implications for the sustainability of biomass production on salt-affected land in more detail. Moreover, future policies on biodiversity restoration and conservation and on forestry can lead to a reduction in the available land area. Combined with increased labour and land costs, this can lead to a reduction of the technical and economic bioenergy potential of salt-affected soils. Another aspect influencing the future extent and availability of salt-affected land for biosaline forestry is climate change. Climate change can lead to either moderation or acceleration of soil salinization and sodification depending on local conditions such as groundwater depth and quality (Munns *et al.*, 1999).

Data availability for determining the yields on salt-affected land, particularly for defining the tree requirements and the constraint-free yields, is a limiting factor. For example, the rating for salinity is based on salinity curves for the juvenile stage of the trees because of the limited availability of salinity curves for tree growth in later stages. However, since trees are generally more susceptible to salts in the juvenile stage than in later stages (Lambert and Turner, 2000), applying the juvenile curve results in lower calculated yields than what is potentially possible. Given that this is the most important variable for saline soils, more work on salinity (and sodicity) curves for later stages is required in order to determine the effect on yields and potential. For the tropical yield model, three land characteristics used in Vashev *et al.*’s (2010) model could not be accounted for in this global study due to the lack of global datasets needed to map them. These are flooding, soil depth, and groundwater depth. Groundwater depth is particularly crucial for tree biomass production in salt-affected environments (Vashev *et al.*, 2010) because salt-affected soils in (sub)tropical regions occur primarily in semi-arid or arid regions where tree growth depends more on groundwater than on precipitation. Therefore, in areas with water tables close to the surface, this study underestimates yields and, consequently, potentials. Future research could further address this shortcoming by, for example, generating a simple global groundwater indicator map and applying it to the global model. Such a map may be generated by combining existing information from geomorphologic maps and drainage network maps. However, this would still only be an approximation of global groundwater levels; more reliable groundwater maps are desirable in the long run. Constraint-free yields for the (sub)tropical tree species are approximated by the highest yield recorded in the literature because constraint-free yield data is not available. This approach underestimates the constraint-free yield and, thereby, results in conservative estimates of actual yields.

Management specific to (different types and severity levels of) salt-affected soils is not included in this study because of the limited data on the precise effects of certain management techniques and of above and belowground biomass growth on the soil characteristics and, thereby, on the yields. Although an increase in yields and technical potential is likely as a result of improved management, it is unclear whether the economic potential also increases. This is because additional management raises per-hectare production costs. Furthermore, using a generalized forestry production system to estimate the biomass yields (and production costs) ignores the impact of differences in management requirements for different species and soil and climate conditions. The impact of these aspects on the yield and production costs should be a central topic for further research.

5.6 CONCLUSIONS

The results of this analysis indicate that salt-affected soils cover approximately 1.1 Gha worldwide and that biomass production on these soils could make a significant contribution to global and regional (bio-)energy demand. The technical potential was calculated to be 56 EJ y^{-1}, or 11% of the current global primary energy consumption. A significant part of the technical potential comes from agricultural land, and its conversion to biomass production may not be considered sustainable. If current agricultural land is excluded, the technical potential decreases to 42 EJ y^{-1}. The analysis of current land use/cover of salt-affected soils also indicates that the lowest production costs and largest potentials are found on land that is currently under shrub and herbaceous cover. However, land from this category is often used for livestock grazing and may therefore only partly be available for biomass production, although agroforestry systems that combine livestock grazing (or agricultural crop production) and biomass production are possible and may actually prevent further salinization/sodification of the land. In order to avoid competition with feed/forage production, future assessments must investigate this topic more carefully.

The economic potential of biomass production on salt-affected soils amounts to 21 EJ y^{-1} at production costs of 2 € GJ^{-1} or less, which is equivalent to 4% of current global energy consumption. Global cost supply curves by severity of salt-affectedness confirm that the largest share of potential comes from the least salt-affected soils. Of the 21 EJ y^{-1} at production costs of 2 € GJ^{-1} or less, 19 EJ y^{-1} are from slightly and moderately salt-affected soils while only 2 EJ y^{-1} are from highly or extremely salt-affected soils. This trend is even more extreme for the economic potential at production costs of 5 € GJ^{-1} or less, where slightly and moderately affected soils account for 93% of the potential.

This study presents an initial assessment of global bioenergy potential from salt-affected soils. Several aspects require additional research. Future research in the field of biosaline forestry should focus on the current use of salt-affected land, on how management affects yields and production costs, and on how biosaline forestry (and agroforestry) can be promoted. In addition, biosaline forestry has numerous additional benefits such as the potential to improve soil conditions, generate income from previously low-productive or unproductive land, and soil carbon sequestration. These are important additional reasons for investigating and investing in biosaline forestry.

5.7 ACKNOWLEDGEMENTS

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5.8 APPENDIX 1: GLOBAL TECHNICAL BIOMASS PRODUCTION POTENTIAL, BY TYPE AND SEVERITY OF SALT-AFFECTEDNESS, LAND USE/COVER, AND REGIONS

Table 5.9: Global technical biomass production potential, by type and severity of salt-affectedness, land use/cover, and regions (in PJ y^{-1})

Type	**Severity**	**Land use/ cover**	**Canada**	**F USSR**	**W Europe**	**USA**	**E Europe**	**E Asia**	**Japan**	**M East**	**S Asia**	**N Africa**	**C America**	**SE Asia**	**W Africa**	**Oceania**	**E Africa**	**S America**	**S Africa**	**Total**
saline	slight	agriculture	0	2344	1	25	1	246	0	135	126	2	1	0	0	453	21	982	27	5502
		shrub / herbaceous cover	0	3305	0	204	3	129	0	511	241	267	73	0	0	122	661	354	812	21832
		bare areas	0	579	0	0	0	96	0	506	103	139	0	0	0	82	69	15	1	1590
		total	*0*	*6227*	*1*	*206*	*4*	*163*	*0*	*115*	*160*	*409*	*75*	*0*	*0*	*128*	*751*	*135*	*840*	*28924*
	moderate	agriculture	0	28	0	0	0	5	0	12	0	1	12	0	0	0	56	4	13	132
		shrub / herbaceous cover	0	57	0	0	0	22	0	113	0	28	143	0	0	0	345	107	488	1303
		bare areas	0	61	0	0	0	30	0	126	0	69	0	0	0	0	71	31	0	388
		total	*0*	*146*	*0*	*0*	*0*	*57*	*0*	*252*	*0*	*98*	*155*	*0*	*0*	*0*	*471*	*142*	*501*	*1823*
	very	agriculture	0	1	0	0	0	7	0	0	0	0	0	0	0	0	0	0	0	8
		shrub / herbaceous cover	0	1	0	0	0	2	0	0	0	0	0	0	0	0	0	0	19	22
		bare areas	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	1
		total	*0*	*1*	*0*	*0*	*0*	*9*	*0*	*0*	*0*	*0*	*0*	*0*	*0*	*0*	*0*	*0*	*20*	*31*
	extreme	agriculture	0	4	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	5
		shrub / herbaceous cover	0	7	0	0	0	0	0	0	0	0	0	0	0	0	1	2	0	9
		bare areas	0	9	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	9
		total	*0*	*19*	*0*	*0*	*0*	*0*	*0*	*0*	*0*	*0*	*0*	*0*	*0*	*0*	*1*	*4*	*0*	*23*
	total		**0**	**6394**	**1**	**206**	**4**	**170**	**0**	**140**	**160**	**506**	**229**	**0**	**0**	**128**	**1223**	**149**	**136**	**30800**
sodic	slight	agriculture	237	233	12	166	38	17	0	0	61	107	1	367	39	348	72	410	135	2243
		shrub / herbaceous cover	245	668	1	237	0	40	0	0	10	33	0	44	69	931	2087	300	132	4796
		bare areas	0	403	0	0	0	34	0	0	0	0	0	0	4	91	5	27	0	564
		total	*482*	*1304*	*13*	*403*	*38*	*90*	*0*	*0*	*70*	*140*	*1*	*411*	*112*	*137*	*2164*	*737*	*267*	*7603*
	moderate	agriculture	117	806	0	187	54	378	0	1	157	212	0	0	221	860	224	134	18	4583
		shrub / herbaceous cover	61	855	0	260	2	129	0	3	18	65	0	0	179	501	605	949	80	8224
		bare areas	0	238	0	0	0	3	0	0	3	0	0	0	0	90	0	6	0	341
		total	*178*	*1899*	*0*	*447*	*57*	*509*	*0*	*4*	*178*	*277*	*0*	*0*	*400*	*596*	*829*	*230*	*98*	*13147*
	very	agriculture	0	0	0	0	0	0	0	0	0	0	0	0	0	0	17	480	0	497
		shrub / herbaceous cover	0	0	0	0	0	0	0	0	0	3	0	0	0	0	163	128	0	294
		bare areas	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	24	0	24
		total	*0*	*0*	*0*	*0*	*0*	*0*	*0*	*0*	*0*	*3*	*0*	*0*	*0*	*0*	*179*	*632*	*0*	*814*

Type	Severity	Land use/ cover	Canada	F USSR	W Europe	USA	E Europe	E Asia	Japan	M East	S Asia	N Africa	C America	SE Asia	W Africa	Oceania	E Africa	S America	S Africa	Total
	extreme	agriculture	0	0	0	0	20	0	0	0	0	0	0	36	0	0	8	11	2	78
		shrub / herbaceous cover	0	0	0	0	1	0	0	0	0	8	0	3	0	0	14	3	39	69
		bare areas	0	0	0	0	0	0	0	0	0	12	0	0	0	0	0	2	0	14
		total	*0*	*0*	*0*	*0*	*21*	*0*	*0*	*0*	*0*	*20*	*0*	*39*	*0*	*0*	*22*	*16*	*42*	*160*
	total		**661**	**3204**	**13**	**849**	**116**	**600**	**0**	**4**	**248**	**439**	**1**	**450**	**511**	**733**	**3195**	**368**	**407**	**21725**
saline-sodic	slight	agriculture	0	0	0	0	0	0	0	0	0	0	0	0	2	0	109	0	0	111
		shrub / herbaceous cover	0	0	0	0	0	0	0	0	0	0	0	0	0	0	154	0	0	154
		bare areas	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	1
		total	*0*	*0*	*0*	*0*	*0*	*0*	*0*	*0*	*0*	*0*	*0*	*0*	*3*	*0*	*263*	*0*	*0*	*267*
	moderate	agriculture	0	20	8	0	9	35	0	0	0	1	0	0	0	0	9	6	1	88
		shrub / herbaceous cover	0	21	0	0	0	8	0	0	0	27	0	0	0	0	167	350	51	625
		bare areas	0	5	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	6
		total	*0*	*46*	*8*	*0*	*9*	*43*	*0*	*0*	*0*	*28*	*0*	*0*	*0*	*0*	*176*	*357*	*52*	*718*
	very	agriculture	2	0	0	0	1	0	0	22	759	1	0	0	47	0	0	0	7	840
		shrub / herbaceous cover	1	0	0	15	1	2	0	40	24	18	0	0	12	0	47	0	13	172
		bare areas	0	0	0	0	0	0	0	48	3	11	0	0	5	0	0	0	0	66
		total	*3*	*0*	*0*	*15*	*1*	*2*	*0*	*111*	*786*	*30*	*0*	*0*	*64*	*0*	*47*	*0*	*20*	*1078*
	extreme	agriculture	1	80	14	3	19	38	0	80	85	30	18	76	58	0	11	1	42	557
		shrub / herbaceous cover	0	168	1	26	5	21	0	117	14	61	12	2	165	48	32	3	79	755
		bare areas	0	126	0	0	0	3	0	180	9	18	0	0	3	0	0	1	7	348
		total	*1*	*375*	*15*	*28*	*25*	*62*	*0*	*378*	*108*	*109*	*30*	*78*	*226*	*48*	*43*	*6*	*128*	*1660*
	total		**5**	**420**	**23**	**43**	**35**	**107**	**0**	**488**	**894**	**168**	**30**	**78**	**293**	**48**	**529**	**363**	**199**	**3723**
Total			**665**	**1001**	**37**	**296**	**155**	**240**	**0**	**189**	**275**	**111**	**260**	**528**	**804**	**201**	**4947**	**554**	**196**	**56248**

Chapter 6

The economic performance and GHG balance of biomass production in (agro)forestry systems on different types of salt-affected soils in South Asia

Abstract: This chapter assesses the economic performance (*i.e.* net present value (NPV) and production costs) and environmental performance (primarily the impact on greenhouse gas emissions) of different biosaline (agro)forestry systems in three cases in South Asia. The economic impact of trading carbon credits generated by biosaline (agro)forestry is also assessed as a potential additional source of income. The NPV at a discount rate of 10% is 1.0 k€ ha^{-1} for a rice-tree agroforestry system on saline soils in coastal Bangladesh (case study 1); 4.8 k€ ha^{-1} for a rice-wheat-tree agroforestry system on sodic/saline-sodic soils in Haryana, India (case study 2); and 2.8 k€ ha^{-1} for a compact tree plantation on saline-sodic soils in Punjab province of Pakistan (case study 3). The GHG balance of the three systems shows carbon sequestration rates of 16 t CO_2-eq. ha^{-1} in Bangladesh, 26 t CO_2-eq. ha^{-1} in India, and 96 t CO_2-eq. ha^{-1} in Pakistan. This translates into economic values that increase the NPV by 3-80% in case study 1, 1-14% in case study 2, and 9-129% in case study 3, depending on the carbon credit price assumed in this study (1-15 € t CO_2-eq.). Although the NPV is positive in all three cases, the analysis indicates that the economic performance strongly depends on the type and severity of salt-affectedness (which affect the type and setup of the agroforestry system and the tree species), tree rotation length, the markets for wood products, the possibility of trading carbon credits, and the discount rate. A simple extrapolation of the results suggests a technical bioenergy production potential from biosaline (agro)forestry of 1.2 EJ y^{-1} (approximately 4% of the total current primary energy consumption of the three countries) and a climate change mitigation potential of 43 Mt CO_2-eq. that could be sequestered annually (approximately 3% of the three countries' GHG emissions related to energy consumption).

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6.1 INTRODUCTION

Salt-affected soils are an important category of degraded soils, both with respect to their large global extent and the severe effects that soil salinity and sodicity have on agricultural productivity. Worldwide, approximately 1.1 billion hectares of land are salt-affected (Wicke *et al.*, submitted; Chapter 5 of this thesis), of which approximately 76 million hectares (Mha) are affected by human-induced salinization and sodification (Oldeman *et al.*, 1991). Furthermore, as a result of improper irrigation management, salinization and sodification of land continue to occur at an estimated rate of between 0.25 and 0.5 Mha each year (FAO, 2000). In South Asia, salt-affected soils and their continuous expansion are a particularly important concern because already-scarce land resources are facing rapidly increasing demands for food, feed, and fuel. The large extent of salt-affected soils is all the more problematic given the fact that soil salinity and sodicity greatly reduce agricultural productivity. There are three types of salt-affected soils (saline, sodic, and saline-sodic soils), each of which affects agricultural productivity differently (US Salinity Laboratory, 1954; Abrol *et al.*, 1988; Lamond and Whitney, 1992; Ghassemi *et al.*, 1995). Saline soils contain high levels of soluble salts, which impede plant growth by increasing osmotic pressure of the soil solution and, in turn, hampering water extraction by plant roots (the osmotic effect). In addition, increased chloride and sodium ion concentrations in the plant can be toxic, causing cell injury and growth reduction (the specific ion effect). Sodic soils are characterized by high levels of exchangeable sodium but low overall salt amounts. Soil sodicity affects plant growth by deteriorating the physical properties of the soil, primarily by increasing soil dispersion, which causes reduced infiltration, reduced hydraulic conductivity, and surface crusting. However, the osmotic effect mentioned above can also be triggered in sodic soils. Saline-sodic soils are characterized by both an excessive amount of soluble salts and high levels of exchangeable sodium and are thus affected by the adverse properties of both saline and sodic soils. In addition, the often co-existing waterlogging (both a cause and a result of the soil salinity/sodicity) reduces oxygen availability in the soil, which further slows growth (Barrett-Lennard, 2003).

Conventional agriculture on severely salt-affected sites is generally not economically viable because agricultural crop yields are low and physical remediation of the salts is often prohibitively expensive for most farmers, including those in South Asia (Ghaly, 2002; Qadir *et al.*, 2002; Qadir and Oster, 2004). However, on these sites biosaline forestry and agroforestry systems may be an alternative land use option. This is because some tree species are less susceptible to soil salinity/sodicity, and their cultivation can help regenerate these soils (Singh *et al.*, 1994; Singh, 1995; Bell, 1999; Lambert and Turner, 2000). Examples of species tolerant to soil salinity, soil sodicity, or both are *Acacia nilotica*, *Eucalyptus camaldulensis*, *Eucalyptus tereticornis*, and *Prosopis juliflora* (US National Research Council, 1990; Marcar and Crawford, 2004). Some tree species have adapted to waterlogging conditions by developing root air channels (aerenchyma) and adventitious (nodal) roots. Examples are *Casuarina obesa*, *Tamarix aphylla* and *Eucalyptus camaldulensis* (Barrett-Lennard, 2002; Barrett-

Lennard, 2003).

Although many examples of research on biosaline agroforestry and forestry systems (herein after (agro)forestry systems) in South Asia and worldwide exist (Singh *et al.*, 1988; Ahmed, 1991; Qureshi *et al.*, 1993; Singh *et al.*, 1994; Singh, 1995; Singh *et al.*, 1997; Bell, 1999; Lefroy and Stirzaker, 1999; Kaur *et al.*, 2002a; Qadir and Oster, 2002; Marcar and Crawford, 2004; Zhang *et al.*, 2004; Masters *et al.*, 2007), only a few studies have evaluated the economic performance of such systems (Bose and Bandoyopadhyay, 1986; Ahmed, 1991; Singh *et al.*, 1994; Singh *et al.*, 1997; Stille *et al.*, submitted). These studies focused on sodic soils and found agroforestry systems to be an economically viable land use option. However, little is known about the economic performance of biosaline (agro)forestry on other types of salt-affected soils in South Asia such as saline soils, saline-sodic soils and waterlogged salt-affected soils. These may perform differently from sodic soils as a result of different management systems, particularly the establishment activities and species planted.

With respect to the environmental performance of biosaline agroforestry, previous studies have focused on the ameliorative effects of trees on soil salinity/sodicity and on soil organic carbon content (*e.g.* Singh et al (1988), Kaur *et al.* (2002b), Qadir and Oster (2002), Lal (2009)and Wong *et al.* (2009)), but have not studied other environmental impacts. In particular, the greenhouse gas (GHG) balance of such systems has not been assessed despite their potential to sequester carbon through revegetating degraded sites and the potential economic benefits from trading carbon credits from biosaline (agro)forestry projects. An analysis of both the economic and environmental performance of these systems is important for a better understanding of the economic potential of bioenergy from salt-affected soils (Wicke *et al.*, submitted; Chapter 5 of this thesis) and the GHG mitigation potential of biosaline (agro)forestry in South Asia.

Given these considerations, the main objective of the present study is to assess the economic and environmental performance of biosaline (agro)forestry on different types of salt-affected soils in South Asia. The economic performance is assessed by studying the net present value (NPV) and the production costs. The environmental performance is assessed by studying the GHG emissions of biosaline (agro)forestry. In addition, by literature research, environmental opportunities and risks of biosaline (agro)forestry systems in terms of biodiversity, water, and soil conditions are evaluated. The economic impact of trading carbon credits generated by biosaline (agro)forestry is also assessed as a potential additional source of income.

In order to capture the effect of different site conditions and production systems, three case studies were executed. Each case study analyzes a biosaline (agro)forestry system in a different setting: 1) a rice-tree agroforestry plantation on coastal saline soils in Bangladesh, 2) a rice-wheat-tree agroforestry plantation on waterlogged, salt-affected soils in India, and 3) a forestry plantation on saline-sodic soils in Pakistan.

This chapter is organized as follows. Section 6.2 describes the soil and climate conditions, the management system, and the biomass applications for each of the three cases. In section 6.3, the methods applied for assessing the economic and environmental performance of biosaline (agro)forestry are explained. Section 6.4 presents input data for the analyses. Section 6.5 presents the results of the analyses. This is followed in Section 6.6 by a discussion on data availability and the feasibility of carbon credit trading from biosaline agroforestry. Section 6.7 presents the study's final conclusions.

6.2 CASE STUDIES

Three case studies of biosaline (agro)forestry in South Asia were assessed (Figure 6.1). Case study 1 (Bangladesh) has a humid, monsoonal climate and soil salinity is induced by sea water intrusion, while case study 2 (India) and 3 (Pakistan) have a semi-arid monsoonal climate and irrigation-induced soil salinity/sodicity problems. These climate and soil conditions are representative for many salt-affected land areas in South Asia. An overview of these cases and their main characteristics are given in Table 6.1. For a more detailed description of each case, see Appendix 6.1.

Table 6.1: The main characteristics of the three case studies of biosaline (agro)forestry in South Asia

	Case study 1	**Case study 2**	**Case study 3**
Location	coastal belt, Bangladesh	Haryana state, India	Punjab province, Pakistan
Soil conditions			
Type of salt-affectedness	saline	saline-sodic topsoil, sodic subsoils, waterlogged	saline-sodic
Severity [a]	ranges from slight to extreme	ranges from slight to moderate	extreme
(Agro)forestry system	alley cropping [b]	alley cropping [b]	compact tree plantation
Tree species	*Acacia nilotica, Eucalyptus camaldulensis*	*Eucalyptus tereticornis*	*Acacia nilotica*
Agricultural crops	rice	rice and wheat	-
Share of land used for agricultural crop (%)	96	96	0
Tree density (trees ha^{-1})	200	200	1730
Lifetime of plantation (years)	20	15	10
Rotations	2 to 4 (depending on biomass application)	3	1 (thinning in year 4 and 6)
Biomass applications	fuelwood, timber	fuelwood, charcoal, timber	fuelwood, timber

a – Severity levels are defined according to the classification by the US Salinity Laboratory (1954)

b – Alley cropping is defined as trees planted in single or grouped rows and agricultural crops in wide alleys between the tree rows (Nair, Kumar and Nair, 2009).

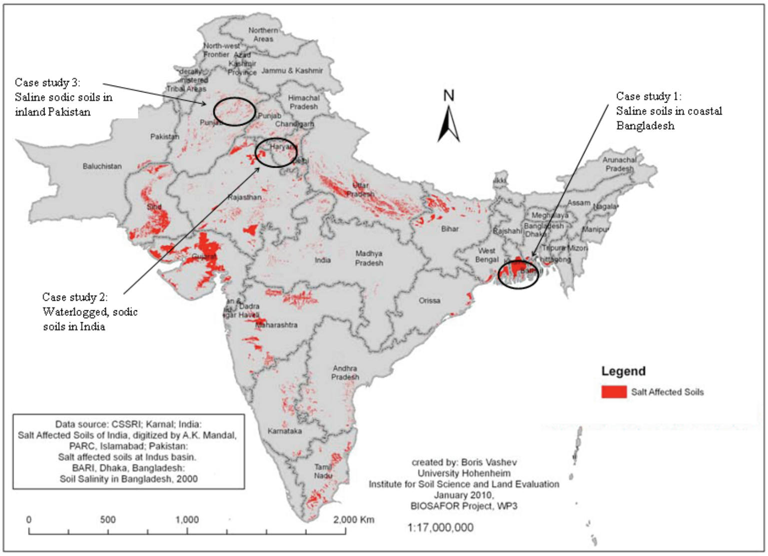

Figure 6.1: Distribution of salt-affected soils in Bangladesh, India and Pakistan (Vashev *et al.*, 2010) and location of case study 1, 2, and 3

6.3 METHODOLOGY

6.3.1 Net present value

The NPV shows how the initial investments required for establishing biosaline (agro)forestry systems compare to the benefits received by the farmer at a later point in time. The NPV of biomass production is calculated as follows

$$NPV = \sum_{i=1}^{n} \frac{B_i - C_i}{(1+r)^i} \qquad (9)$$

where B_i – benefits in year i (€ ha^{-1}); C_i – costs in year i (including initial investments in the first year) (€ ha^{-1}); r – real discount rate (%); n – lifetime of project (years).

Carbon credits from biosaline (agro)forestry are determined based on the GHG balance (Section 6.3.3) and are assumed to be sold in one of the various carbon markets. The value of carbon sequestration from biosaline (agro)forestry is integrated in the NPV as an additional benefit of biosaline (agro)forestry. Thus, in equation 9 above, the benefits (B_i) also account for the economic value of carbon sequestration. As carbon sequestration occurs throughout the plantation lifetime (assuming a new equilibrium is not reached before), the benefits are assumed to be received annually.

6.3.2 Production costs

Biomass from biosaline (agro)forestry systems may be used for various applications. To determine the competitiveness of different applications of biosaline biomass with conventional biomass, this study assesses the production costs of fuelwood and timber (as well as charcoal for case study 2, India) and compares them to the market prices of these products in the case study areas. Production costs are determined by accounting for inter-annual differences in costs and benefits (van den Broek *et al.*, 2000; Smeets *et al.*, 2009c):

$$P_{\cos t} = \frac{\sum_{i=1}^{n}\left(\frac{C_i - B_{bp,i}}{(1+r)^i}\right)}{\sum_{i=1}^{n}\frac{Y_i}{(1+r)^i}} \times CE + C_{proc} \tag{10}$$

where P_{cost} – cost of biomass production (€ per tonne dry matter (t dm)); C_i – costs in year *i* (€ ha^{-1}); $B_{bp,\,i}$ – benefits from by-products in year *i* (€ ha^{-1}); *r* – real discount rate (%); Y_i - yield of wood in year *I* (t ha^{-1}); *CE* – conversion efficiency (t input t^{-1} output); C_{proc} – costs of processing (including transportation) (€ (t dm)$^{-1}$); *n* - lifetime of the project (years).

For systems that produce multiple products, partitioning of cost items is required. If there is one main product and one or more by-products (*e.g.* pods in the case of *Acacia nilotica* in Bangladesh), the benefits of the by-products are subtracted from the production costs as described by equation 10. If multiple co-products are produced (*e.g.* fuelwood and timber), cost items are first partitioned to the respective product whenever possible. If this is not possible (*e.g.* for cost items that are applied for both products), the costs are allocated based on their economic value.

6.3.3 Determining GHG emissions and exploring other environmental impacts of biosaline (agro)forestry

A GHG balance for biomass production from each of the three case studies is constructed based on emissions from all activities on the plantation, *i.e.* land use change, establishment and operation of the plantation, harvest, and in-field transportation. Carbon emissions and sequestration from land use change account for carbon stock changes in belowground biomass, litter, and soil, and are calculated according to the IPCC Guidelines for National Greenhouse Gas Inventories (IPCC, 2006). Aboveground biomass is not accounted for because its consumption for energetic purposes releases the sequestered carbon (see also Section 6.6.2). Carbon emissions from changes in belowground biomass and litter are determined by subtracting the time-averaged carbon stock of the plantation from the carbon stock of the original vegetation. The net carbon emissions are then allocated equally over the lifetime of the (agro)forestry plantation. Average annual soil carbon sequestration for afforestation of salt-affected soils is estimated based on available literature. Indirect emissions from land use change, *i.e.* through the displacement of previous land use, are not accounted for in this study. This is because either the case study sites are so severely degraded that land was lying idle before conversion to biosaline forestry (case study 3) or the addition of trees helps improve soil and water conditions so that yields of the intercrops are increased, thereby compensating for the reduction in the area planted (case study 1 and 2). Emissions from activities or inputs to the agroforestry system that cannot clearly be assigned either to the agricultural crop or to the tree cultivation are allocated based on the economic value of the two crops. The three most important GHGs (carbon dioxide (CO_2), methane (CH_4), and nitrous oxide (N_2O)) are included and expressed as CO_2 equivalents (CO_2-eq.). Other main GHGs (hydrofluorocarbons, perfluorocarbons, sulphur hexa-fluoride) are not taken into account as they are insignificant in bioenergy production chains. In addition to the GHG balance, a simple estimation of avoided GHG emissions is made considering the co-firing of biosaline biomass with coal for electricity production.

Other environmental impacts of biosaline (agro)forestry are investigated on an explorative level only in terms of opportunities and risks. Aspects discussed include the quantity and variability of biodiversity, potential invasiveness and weediness of salt-tolerant tree species, soil salinity/sodicity control, soil carbon sequestration, other improvements in soil conditions, waterlogging control, competition for water, and impacts of irrigation.

6.4 INPUT DATA

The majority of the cost data for the case studies was obtained from the local research institutes Bangladesh Agricultural Research Institute (BARI, Bangladesh), Central Soil Salinity Research Institute (CSSRI, India), and Nuclear Institute for Agriculture and Biology (NIAB, Pakistan). In addition, cost data were collected by interviewing farmers, wood traders, and forestry officers near the case study sites in April 2008 and in May to June 2009. Whenever data from the case studies were insufficient, additional data were gathered from scientific literature, research reports and management plans. Input data and their sources are described below and in Appendix 6.2.

6.4.1 Economic performance

For case study 1 (Bangladesh), harvested biomass yields from *Acacia nilotica* and *Eucalyptus camaldulensis* are estimated for different severity levels by applying the species-specific relationship of mean annual biomass increment to soil salinity as determined by Vashev *et al.* (2010) from a collection of biomass and soil data from 15 case studies in India, Pakistan, and Bangladesh. This results in *Acacia nilotica* yields ranging from 3.0 t ha^{-1} y^{-1} at slightly saline sites to 2.3 t ha^{-1} y^{-1} at extremely saline sites and in *Eucalyptus camaldulensis* yields ranging from 4.6 t ha^{-1} y^{-1} at slightly saline sites to 2.4 t ha^{-1} y^{-1} at extremely saline sites. *Acacia nilotica* pods are produced at a rate of 25 kg $tree^{-1}$ y^{-1} starting in the fifth year (Viswanath *et al.*, 2001). In earlier years, the trees are likely to produce some pods but at lower rates. This is not included in the present study as no information is available on the amounts. A rice yield of 2.5 to 3.0 t ha^{-1} y^{-1} is typical for coastal soils during the monsoon season (Haque, 2006); the average of this range is applied in this study. Annual rice production costs in Bangladesh are 325 € ha^{-1} (Department of Agricultural Extension, 2005).

Case study 2 (India) has a harvested biomass yield of 4.4 t dm ha^{-1} y^{-1}. Rice yields at the case study are reported to have increased by 30% to 3.9 t ha^{-1} for wheat grown in the winter season and 2.2 t ha^{-1} for rice in the monsoon season after the lowering of the water table. The rice and wheat yields are assumed to reach this level gradually in six years. Annual rice and wheat production costs in Haryana are 250 € ha^{-1} and 287 € ha^{-1}, respectively (HAU, 2005). Reclamation of waterlogged, saline-sodic/sodic soils in Haryana is done by installing a surface and subsurface drainage system (to reduce waterlogging), applying large quantities of gypsum and then flooding the field (to leach out the salts). The costs of installing a surface and subsurface drainage system are approximately 560 € ha^{-1}, and the costs of reclaiming sodic or saline-sodic soils by applying gypsum and flooding the field amount to approximately 300 € ha^{-1} (Stille, 2009).

In case study 3 (Pakistan), 16 t ha^{-1} are harvested from thinning the stand in year 4, 38 t ha^{-1} are harvested from thinning the stand in the sixth year, and 52 t ha^{-1} are harvested at the final harvest in the tenth year.

Other input data for the economic performance analysis are presented in Table 6.4 in Appendix 6.2.

6.4.2 GHG emissions

GHG emissions from land use change are assessed based on previous land use and tree biomass productivity from the case studies. Previous land use in the case studies is rice production in Bangladesh, rice-wheat production in India, and fallow land with low vegetative productivity in Pakistan. As a result, aboveground biomass prior to conversion to biosaline (agro)forestry is assumed to be zero for Bangladesh and India, while a total of 1 t C ha^{-1} is assumed for uncultivated salt-affected land in Pakistan based on data given in Gibbs *et al.* (2008) for degraded land. Soil carbon sequestration by afforestation of salt-affected soils is estimated by Lal (2004b) to be in the range of 0.2 to 0.5 t C ha^{-1} y^{-1}. The average (0.35 t C ha^{-1} y^{-1}) of this range is used in this study. Carbon uptake by woody biomass growth is accounted for by applying biomass yields from the case studies (Table 6.4). Belowground biomass is determined as a percentage of aboveground biomass (Kaur *et al.*, 2002b; IPCC, 2006; Ram *et al.*, 2008). Carbon stocks in litter and dead wood in the reference system and the agroforestry system are estimated based on default data from IPCC (2006). Input data for emissions from fossil diesel application are a diesel emission factor of 74.1 g CO_2 MJ $_{LHV}$ and a diesel energy content of 43 GJ t^{-1} (IPCC, 2006). Application rates of organic and synthetic fertilizers in the case studies are presented in Table 6.4. An emission factor for urea production of 1326 g CO_2-eq. (kg N produced)$^{-1}$ (Wood and Cowie, 2004) is applied, and the default values for direct and indirect N_2O emission factors from fertilizer application are taken from IPCC (2006). GHG emissions from the production of gypsum for use in the case studies in India and Pakistan are estimated at 0.01 t CO_2-eq. per tonne gypsum (Ecofys *et al.*, 2009). The application of gypsum has, in some cases, been shown to reduce CH_4 emissions of rice production (Wassmann *et al.*, 2004), but it is not known whether this is a general effect, and this factor is therefore not included in the analysis. The estimation of GHG emissions avoided by co-firing biosaline biomass with coal for electricity production assumes additional emissions for the transportation and pre-treatment of biomass of 7 g CO_2-eq. MJ^{-1}; an average electric efficiency of 30% for coal power generation in South Asia (Bhattacharya, 2006); GHG emissions of coal combustion of 95 g CO_2-eq. MJ^{-1} (IPCC, 2006); and indirect energy requirements for coal extraction, transportation, and storage of 7% (Blok, 2006).

The price of carbon credits varies strongly depending on the market (compliance vs. voluntary market), the type of project (such as agricultural soil sequestration or afforestation/reforestation projects), the economic outlook, and regulatory uncertainties (Capoor and Ambrosi, 2009; Ecosystem Marketplace and New Carbon Finance, 2009). In order to reflect the large variation in prices and the possible impact on the economic performance of biosaline (agro)forestry, the present study applies three carbon credit prices: 1, 5, and 15 € t^{-1} CO_2-eq.

6.5 RESULTS

6.5.1 Economic performance and GHG emissions

6.5.1.1 Case study 1: Coastal saline soils in Bangladesh

The NPV of the (agro)forestry system in coastal saline soils in Bangladesh depends strongly on the severity of the soil salinity, the tree species cultivated, and the tree rotation length. On slightly, moderately, and highly saline land, the NPV of the *Eucalyptus camaldulensis* timber production is the highest (1.4 k€ ha^{-1}, 1.3 k€ ha^{-1}, and 1.0 k€ ha^{-1}, respectively), while in extremely saline areas the NPV of the production of *Acacia nilotica* timber is the highest (0.9 k€ ha^{-1}) (Figure 6.2). This is the result of *Acacia nilotica* being more salt-tolerant than *Eucalyptus camaldulensis* but *Eucalyptus camaldulensis* having a higher productivity than *Acacia nilotica* at lower soil salinity. At all severity levels and for both species, timber production results in higher NPV than fuelwood production, primarily due to higher market prices for timber. For comparison, on non-saline soils where three rice cropping seasons are possible, the NPV over the same period is 1.6 k€ ha^{-1} (based on data from Department of Agricultural Extension (2005)), and on saline soils with only one rice harvest, the NPV over the same period is 0.3 k€ ha^{-1}. This indicates that agroforestry systems on coastal saline soils in Bangladesh are more economically beneficial than one rice harvest and comparable to three cropping seasons at non salt-affected sites.

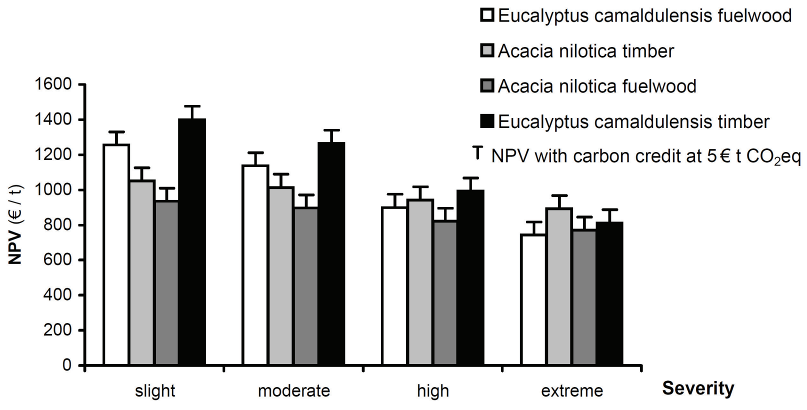

Figure 6.2: NPV of (agro)forestry systems in coastal saline soils of Bangladesh (10% discount rate)

The GHG balance of biomass production from biosaline agroforestry in the coastal saline soils of Bangladesh indicates that this system sequesters carbon at a rate of 20 g CO_2-eq. MJ^{-1} (Table 6.2) as a result of increased carbon stocks in belowground biomass, litter, and, most importantly, the soil. The total carbon sequestration over the lifetime of the plantation is equivalent to 16 t CO_2-eq. ha^{-1}, and its total economic value (in terms of the present value over the lifetime of the plantation) ranges between 0.02 and 0.3 k€ ha^{-1} depending on the carbon credit price assumed (Table 6.2). Assuming a 10% real discount rate, this additional benefit increases the NPV of *Acacia nilotica* fuelwood production by 3% (at a 1 € t^{-1} CO_2-eq. carbon credit price) to 80% (at a 15 € t^{-1} CO_2-eq. carbon credit price) on highly saline soils. The estimation of avoided GHG emissions when co-firing biosaline biomass with coal for electricity production indicates that approximately 115 g CO_2-eq. MJ^{-1} could be avoided for every MJ biomass used in co-firing.

Production costs of fuelwood are 24 € $(t\ dm)^{-1}$ for *Eucalyptus camaldulensis* and 29 € $(t\ dm)^{-1}$ for *Acacia nilotica*, both of which are lower than the average market prices for fuelwood (Table 6.2). Production costs for timber are 124 € $(t\ dm)^{-1}$ for *Eucalyptus camaldulensis* $^{-1}$ and 130 € $(t\ dm)^{-1}$ for *Acacia nilotica*, which are comparable to the upper end of the range in market prices (Table 6.2).

6.5.1.2 Case study 2: Waterlogged, sodic/saline-sodic soils in Haryana, India

The results of the assessment of biosaline (agro)forestry on waterlogged, sodic/saline-sodic soils in Haryana, India show that the NPV of the (agro)forestry biomass production systems is more than three times higher than the NPV of the reference situation (conventional agriculture without amendments) (Figure 6.3). The significantly greater NPV of the strip plantation of trees is due to the additional income from cultivating trees and, most importantly, the additional income from the increased rice and wheat crop yields. The latter is the result of trees extracting groundwater and thereby lowering the groundwater table to a level that does not negatively affect agricultural crop production. Increasing yields is not only important for generating more profits for the farmer, but it also increases food security. The NPV of the agricultural system with drainage and gypsum is also significantly higher than the baseline (Figure 6.3), but slightly lower than the strip plantation. However, a mere 15% reduction in tree biomass yields is enough to make the NPVs of the two systems equivalent. An important aspect of the drainage and chemical amendment system is that its initial investment costs are more than four times higher than those for the establishment of trees.

The GHG balance of biomass production from biosaline agroforestry in the salt-affected soils of India indicates that this system sequesters carbon at a rate of 24 g CO_2-eq. MJ^{-1} (Table 6.2). The total carbon sequestration over the lifetime of the plantation is equivalent to 26 t CO_2-eq. ha^{-1}, and its total economic value (in terms of the present value over the lifetime of the plantation) ranges between 0.05 and 0.7 k€ ha^{-1} depending on the carbon credit price (Table 6.2). Assuming a 10% real discount rate, this additional benefit increases the NPV by between 1% (at a 1 € t^{-1} CO_2-eq. carbon credit price) and 14% (at a 15 € t^{-1} CO_2-eq. carbon credit price). The estimation of GHG avoided emissions when co-firing biosaline biomass with coal for electricity production indicates that approximately 118 g CO_2-eq. MJ^{-1} could be avoided for every MJ biomass used in co-firing.

The cost of production of fuelwood (42 € t^{-1}), timber (120 € t^{-1}), and charcoal (188 € t^{-1}) are considerably lower than market prices, indicating that these biosaline biomass products can be competitive with existing production chains (Table 6.2).

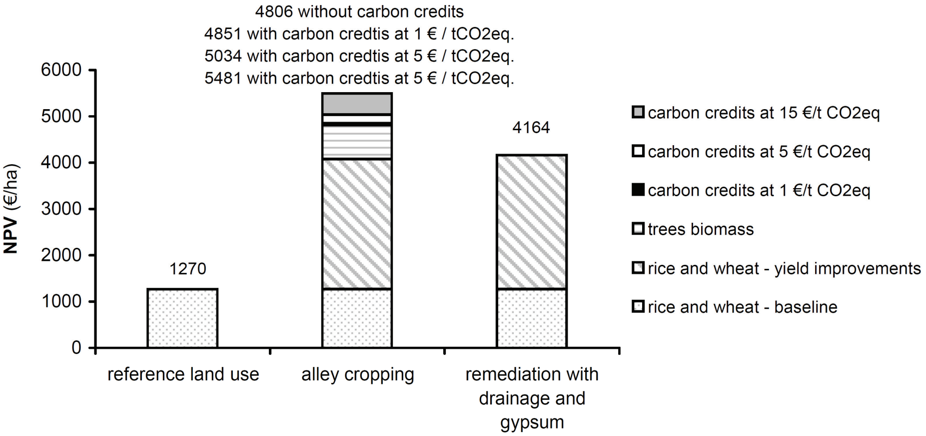

Figure 6.3: NPV of biodrainage agroforestry compared to conventional agriculture and conventional agriculture with drainage in Haryana, India

6.5.1.3 Case study 3: Saline-sodic soils in Pakistan

The compact tree plantation on saline-sodic soils in Pakistan evaluated in this study has a positive NPV even though the land is extremely salt-affected and difficult to use for conventional agriculture (Figure 6.4). The NPV strongly depends on the applied discount rate because financial benefits are received several years after the large initial investments. At a 10% discount rate, the NPV amounts to 2.8 k€ ha^{-1}. Varying the discount rate from 5% to 20% shows NPVs between 5.1 and 0.3 k€ ha^{-1}.

The GHG balance of biomass from the forestry plantation on saline-sodic soils in Pakistan is negative, sequestering carbon at a rate of 9.5 t CO_2-eq. ha^{-1} y^{-1} (equivalent to 46 g CO_2-eq. MJ^{-1} biomass) (Table 6.2) as a result of converting salt-affected land with low productivity into a forestry plantation that sequesters carbon in belowground biomass, litter, and soil. Because of the large sequestration potential, the present value (at 10% discount rate) of carbon sequestration over the lifetime of the plantation ranges from 0.2 k€ ha^{-1} (at a carbon price of 1 € t^{-1} CO_2-eq) to 3.2 k€ ha^{-1} (at a carbon price of 15 € t^{-1} CO_2-eq) (Table 6.2). This leads to an increase in the NPV (at 10% discount rate) of between 9% and 129% compared to NPV without the benefits of trading carbon credits. If biosaline biomass is co-fired with coal for electricity production the net avoided GHG emissions amount to approximately 140 g CO_2-eq. MJ^{-1}.

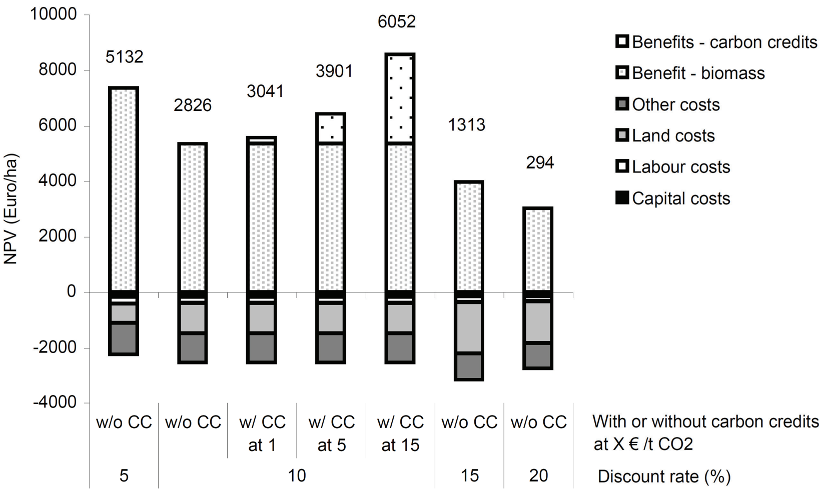

Figure 6.4: NPV of a forestry plantation on saline-sodic soils in Pakistan

Fuelwood production costs range from 54 € t^{-1} (2.8 € GJ^{-1}) at a real discount rate of 5% to 123 € t^{-1} (6.4 € GJ^{-1}) at a real discount rate of 20% (Table 6.2; Figure 6.5). The average fuelwood market price of 53 € t^{-1} is slightly lower than production costs even at a discount rate of 5%. Timber production costs range from 108 € t^{-1} at a real discount rate of 5% to 256 € t^{-1} at a real discount rate of 20% (Table 6.2; Figure 6.5). Production costs are lower than the average market price of 127 € t^{-1} only at a discount rate of 5%. Thus, whether fuelwood and timber from salt-affected soils can compete with market prices depends strongly on the discount rate and the actual market price.

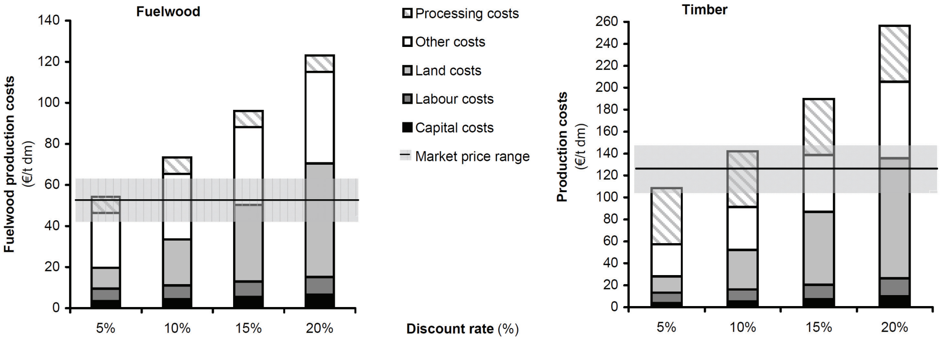

Figure 6.5: Production costs and market prices of timber and fuelwood from forestry plantations on saline-sodic soils in Pakistan

Table 6.2: Overview of economic performance and GHG balances of biosaline (agro)forestry systems in three case studies in South Asia

	Unit	**Case study 1 (Bangladesh)** [a]		**Case study 2 (India)**		**Case study 3 (Pakistan)** [b]	
GHG balance for biomass production							
LUC	g CO$_2$-eq. MJ^{-1}	-21.6		-28.6		-48.1	
Fossil fuels	g CO$_2$-eq. MJ^{-1}	1.0		4.4		2.3	
Agrochemicals	g CO$_2$-eq. MJ^{-1}	0.2		0.2		0.1	
Total	g CO$_2$-eq. MJ^{-1}	-20.4		-24.0		-45.7	
Carbon sequestration	t CO$_2$-eq. ha^{-1} y^{-1}	0.5		1.6		9.5	
Avoided emissions [h]	t CO$_2$-eq. ha^{-1} y^{-1}	5.7		7.0		12.4	
NPV	€ ha^{-1}	992		4806		2826	
Present value of carbon sequestrations (over lifetime of plantation, 10% real discount rate) **at carbon credit price of**							
1 € t^{-1} CO$_2$-eq	€ ha^{-1}	22		48		215	
5 € t^{-1} CO$_2$-eq	€ ha^{-1}	109		241		1075	
15 € t^{-1} CO$_2$-eq	€ ha^{-1}	328		723		3226	
Production costs (*PC*) **vs. (range in) market prices** (*MP*)							
		PC	*MP* [c, f]	*PC*	*MP* [c, e]	*PC*	*MP* [c, d]
Fuelwood	€ (t dm)$^{-1}$	24	33 (27 – 38)	42	88	73	53 (43 – 63)
Charcoal	€ (t dm)$^{-1}$	-	-	188	354		
Timber [c]	€ (t dm)$^{-1}$	124	119 (104 – 134)	120	159	142	127 (107 - 147)[g]

a – Results shown for case study 1 refer to *Eucalyptus camaldulensis* production on highly saline soils;

b – Results of the NPV and production costs analysis for case study 3 refer to a discount rate of 10%;

c – Market prices for fuelwood, charcoal, and timber strongly depend on the tree species. For example, market prices for *Dalbergia sissoo* wood is up to 150% higher than for other species in Bangladesh, while it is nearly three times higher than *Eucalyptus tereticornis* in India. Market prices presented in the table refer to the species grown at the case study site. If species-specific data is not available, the range of market prices is presented;

d – Based on interviews with wood traders in Faisalabad, April 2008;

e – Prices at case study site (Puthi, India) determined from interviews with farmers and wood traders, April 2008;

f – Estimates by forest range officer in Khajura, Bangladesh, April 2008;

g – Range is determined based on market price range of fuelwood in Pakistan and fuelwood and timber in Bangladesh;

h – Avoided emissions when co-firing biomass in coal power plant.

6.5.1.4 Bioenergy and climate change mitigation potentials of biosaline agroforestry in Bangladesh, India, and Pakistan

A simple extrapolation of the results (taking the average yield of the three case studies of 6 t dm ha^{-1} y^{-1} and the salt-affected land area in Bangladesh, India, and Pakistan of approximately 11 Mha) indicates a technical bioenergy production potential from biosaline (agro)forestry of 1.2 EJ y^{-1} for the three countries. This is equivalent to nearly 4% of the total current primary energy consumption of the three countries of 31 EJ y^{-1} (OECD/IEA, 2010c). A breakdown by country indicates that this is 9% for Bangladesh, 3% for India, and 9% for Pakistan. The potential climate change mitigation by biosaline agroforestry in the three countries (taking the average carbon sequestration of the three case studies of 3.9 t CO_2-eq. ha^{-1} y^{-1}) amounts to a total of 43 million tonnes (Mt) CO_2-eq. that could be sequestered annually. This is equivalent to 3% of the three countries' GHG approximately 1,600 Mt CO_2-eq. energy-related emissions in 2008; (OECD/IEA, 2010a). Furthermore, there is an additional avoided GHG emission potential, particularly for India, when the biomass from biosaline (agro)forestry is used to replace coal in electricity generation. This would amount to avoided emissions of another 50 Mt CO_2-eq. (or 3.5 % of India's energy-related CO_2 emissions in 2008; (OECD/IEA, 2010a)).

6.5.2 Exploration of other environmental impacts of biosaline (agro)forestry

In order to provide a general idea of other environmental impacts of biosaline (agro)forestry (in addition to the GHG balance), the following sections explore environmental opportunities and risks (see Table 6.3 for an overview) and provide suggestions for avoiding or minimizing risks.

Table 6.3: Overview of environmental opportunities and risks of biosaline (agro)forestry

Theme	Opportunities	Risks
Biodiversity	- Increasing quantity and variability of biodiversity	- Invasiveness and weediness of salt-tolerant tree species
Soil	- Soil salinity/sodicity control - Soil carbon sequestration - Other improvements in soil conditions	- Deterioration of soil salinity/sodicity as a result of biodrainage in hydrological discharge areas
Water	- Waterlogging control	- Competition for water - Impacts of irrigation

6.5.2.1 Biodiversity

By re-vegetating a degraded site (case study 3, Pakistan) or by diversifying crops (rice and trees in the coastal saline soil case of Bangladesh; rice, wheat, and trees in the case of sodic, waterlogged soils in India), biosaline (agro)forestry plantations may help increase the quantity and variability of biodiversity compared to the reference land use. However, the setup and management of (agro)forestry plantations on salt-affected and other degraded land have a large impact on the level and variability of biodiversity (Dornburg *et al.*, 2010; Plieninger and Gaertner, 2011). Field data on biodiversity levels of salt-affected areas and of reclaimed areas are scarce, and more research is needed on the actual biodiversity of salt-affected land with different severity levels and on the effect of biosaline (agro)forestry on the quantity and variability of biodiversity.

Although biodiversity of salt-affected soils may increase with the introduction of biosaline (agro)forestry, there is also the risk that salt-tolerant (tree) species may become invasive and weedy and spread into neighbouring, non-salt-affected areas. This is especially (but not exclusively) the case for the introduction of exotic species that are potentially more salt-tolerant than local species and which have no natural predators or competitors. In addition, salt-tolerant species are often less susceptible to other stresses (*e.g.* droughts, fires, and waterlogging). Invasion of these species may replace native species and thereby change the ecosystem and biodiversity of non-salt-affected areas. In order to reduce the probability of invasion and weediness, native salt-tolerant plants should be used whenever possible. If exotic species are planted, they must be screened with great care for their invasiveness and weediness potential before introducing them into a new environment (see Carter (1998) for an overview of what such screening should entail). In addition, (agro)forestry plantations with introduced species must also apply appropriate management techniques in order to minimize the risk of invasion. Management must be specific to the tree species because different species have different stress tolerances and seed dispersal mechanisms (see, for example, Pasiecznik (1999) for management options specific to *Prosopis juliflora*).

6.5.2.2 Soils

Biosaline (agro)forestry's most important opportunity lies in controlling soil salinity and sodicity as described in Section 6.1. Many cases have demonstrated agroforestry systems' ability to reduce soil salinity or sodicity (Singh *et al.*, 1994; Bell, 1999; Dagar *et al.*, 2001; Barrett-Lennard, 2002; Kaur *et al.*, 2002b; Kaur *et al.*, 2002a; Qadir and Oster, 2002). However, despite its positive role, biosaline (agro)forestry can also represent a risk for soil salinity/sodicity. Heuperman *et al.* (2002) describe that biodrainage in hydrological discharge areas can result in salt accumulation underneath the plantation. Salinization of groundwater and soils as a result of afforestation can occur because of increased evapotranspiration and groundwater consumption by trees (Jobbágy and Jackson, 2004). However, evidence is limited, and more research is needed on quantifying this effect (particularly in the long term) and on determining the conditions under which this effect occurs (Heuperman *et al.*, 2002).

By increasing biomass growth and reducing other degradation processes (*e.g.* soil erosion, dispersion, and leaching), biosaline (agro)forestry can improve soil conditions. The effect of increased soil organic carbon content as a result of biosaline (agro)forestry has been assessed in various studies. In the literature soil carbon sequestration is estimated to be between 0.1 and 2.9 t C ha^{-1} y^{-1} (Lambert and Turner, 2000; Lal, 2001; Kaur *et al.*, 2002b; Lal, 2004a; Lal, 2004b; Qadir and Oster, 2004; Lal, 2009; Wong *et al.*, 2009). The large variation in sequestration is primarily explained by differences in the soil type, initial soil conditions, climate, and the tree species (as a result of different litter production rates). Even at the lower level of this range, soil carbon sequestration is an important benefit of biosaline (agro)forestry. Other improvements in soil conditions include reducing water erosion (and thereby nutrient losses) through improving water infiltration, reducing impacts by water droplets, intercepting rain and snow, and physically stabilizing soil through roots and leaf litter (Kort *et al.*, 1998). Not only can trees reduce the loss of nutrients, but they can actually increase the supply of nutrients within the rooting zone of crops through fixing biological nitrogen gas (N_2) and retrieving nutrients from below the rooting zone of crops (Buresh and Tian, 1997). Examples of salt-tolerant tree species that are N_2-fixing trees are *Acacia nilotica* and *Prosopis juliflora*.

6.5.2.3 Water

Biosaline (agro)forestry systems may potentially have the positive effect of improving water infiltration and soil moisture retention and may provide an opportunity for improving yields of agricultural crops intolerant to waterlogging. The latter is the result of including trees in the agricultural production system that can help remove excess water and thereby reduce waterlogging, as seen in case study 2 (India). Various studies have confirmed that, when properly implemented, biodrainage systems can lower groundwater tables (for an overview see (Heuperman *et al.*, 2002). However, as mentioned above, Heuperman *et al.* (2002) also indicate that biodrainage in hydrological discharge areas can result in salt accumulation underneath the plantation, and more research is need to clarify this risk. Related to the removal of excess water is also the risk of an exacerbation of water shortages in water-scarce regions due to the additional competition for water (Berndes, 2002). In addition to the direct uptake of groundwater by trees, irrigation of biosaline (agro)forestry will increase the pressure on water resources. Competition for high quality water can be reduced by irrigating with brackish water not useful for other purposes. However, applying brackish irrigation water can cause even more severe soil salinity problems when the amount of irrigation water does not allow for sufficient leaching of the salts. Therefore, prior to establishing biosaline (agro)forestry plantations, local water resources and potential impacts by the (agro)forestry system must be assessed. Further investigating the economic, social, and environmental impacts of increased competition for water is recommended.

6.6 DISCUSSION

6.6.1 Data availability and data quality

The biomass yields and market prices of wood products are important factors in the economic performance analysis. In the present study, biomass yields are estimated based on the relationship of mean annual biomass increment to soil salinity as determined by Vashev *et al.* (2010) for case study 1 (Bangladesh), actual biomass harvested after the first rotation for case study 2 (India), and biomass yield data collected from research plots for case study 3 (Pakistan). However, biomass yields are strongly dependent on local site conditions (*e.g.* type and severity of salt-affectedness, water availability), the tree species, and management practices (*e.g.* fertilizer application, irrigation, and harvest age). Market prices, particularly for fuelwood and charcoal, show a large variation as a result of species-specific energy content, ash content, moisture content, and wood density. But there is also a (large) variation in market prices at different locations as there is no open, transparent market of fuelwood and charcoal in rural areas. For this study, market prices of fuelwood, timber, and charcoal were collected by interviewing wood traders, and forestry officers near the case study sites and whenever possible, a range of market prices was given to show the possible variations.

Another crucial factor in the economic performance analysis of (agro)forestry systems is the discount rate because returns on the large initial investments are obtained over a long period of time. The discount rate may be based on the rate of return for the investors' best alternative investment or the rate paid for the borrowed capital. The interest rate varies per year and source of the loan. Because the value of money generally decreases over time due to inflation, the interest rate is often corrected by taking the real discount rate (*i.e.* the actual discount rate minus rate of inflation) instead of the actual (market) discount rate (Blok, 2006). The literature overview of (real) interest rates in India, Pakistan, and Bangladesh indicates that, especially in Pakistan, the rate strongly varies over time and source of the loan. Therefore, in case study 3 (Pakistan), a discount rate ranging from 5% to 20% was applied. The results indicate that the NPV at 5% is nearly 20 times higher than at 20%. The discount rate is, therefore, a crucial factor in whether (agro)forestry is taken up by a farmer.

Other aspects that affect production costs and biomass yields, and thus the overall economic performance of these systems, are economic development in South Asia and the resulting impacts on, among other things, labour costs; the dissemination of knowledge and skills related to biosaline agroforestry; and the application of sustainable water and soil management practices.

Although it was included in the estimation of net avoided GHG emissions, co-firing biomass with coal for electricity generation was not included in the economic performance analysis due to lack of data. However, co-firing biomass and small-scale biomass gasification from locally grown biomass for electricity generation have the additional benefits of bringing energy self-sufficiency to remote communities and, through the added economic and social benefits of electrification, allowing enhanced rural development (OECD/IEA, 2010b). Further investigation of the economic performance of co-firing and (small scale) gasification of biomass from biosaline (agro)forestry for electricity generation is recommended.

6.6.2 Carbon credits from biosaline (agro)forestry

Several aspects of determining carbon credits and the economic value require discussion. First, soil carbon sequestration rates are not available for the three case studies presented here. However, soil carbon sequestration is an important factor in the GHG balance and is strongly related to local conditions (*e.g.* soil structure) and the setup of the (agro)forestry system (*e.g.* proportion of trees, tree species, and soil management) ((Montagnini and Nair, 2004; Nair *et al.*, 2009); see also Section 6.5.2.2). Considering the range of soil carbon sequestration for Pakistan given in Lal (2004b) of 0.2 to 0.5 t C ha^{-1} y^{-1} results in a decrease of 6% of total carbon sequestration at the lower limit of this range and an increase of 6% at the upper limit. However, other studies indicate even higher soil carbon sequestration rates by agroforestry and forestry plantations on salt-affected soils (see Section 6.5.2.2), in which case the positive effect of carbon sequestration would be enhanced. Carbon sequestration in aboveground biomass is not accounted for in the GHG balance because the sequestered carbon is emitted during its use. However, on average more carbon is stored in aboveground biomass than in the reference land use and accounting for this would further increase the sequestration potential of biosaline (agro)forestry in South Asia.

Second, this study made a simple estimation of net avoided emissions when biosaline biomass is co-fired with coal to produce electricity. However, although co-firing of biomass does take place in India, where coal accounts for approximately 70% of electricity production (OECD/IEA, 2010c), this is not the case for Pakistan and Bangladesh. In these countries, biomass from biosaline (agro)forestry is likely to replace biomass from other, potentially unsustainable, sources. It would, therefore, be more realistic to determine net avoided emissions based on the assumption that deforestation elsewhere is avoided. However, quantifying avoided deforestation and its avoided emissions is only recognized in a few voluntary carbon trading schemes, and carbon credits for avoided deforestation are traded at very low prices (Ecosystem Marketplace and New Carbon Finance, 2009). Future possibilities for accounting for avoided deforestation may be created by the REDD+ mechanism proposed for inclusion in a post-Kyoto climate mitigation strategy. In addition to the indirect land use change of avoided deforestation, there is also another potentially positive indirect land use change as a result of agricultural crop yield increases. In India for example, the introduction of trees results in a lowering of groundwater tables and an increase in rice and wheat yields. This may result in a decrease in the overall amount of land needed for the production of rice and wheat.

Thirdly, this study determined the economic value of carbon sequestration by applying three carbon prices (1, 5, and 15 € t^{-1} CO_2-eq.) based on the variations in current market prices for carbon from different projects and in different carbon markets. However, future prices of 30-50 € t^{-1} CO_2-eq. are also suggested in the literature (see, *e.g.,* the World Energy Outlook 2010 (OECD/IEA, 2010b)), which would double or even triple the economic value of carbon sequestration.

Fourthly, this study did not account for the transaction cost of carbon credit trading when estimating the economic value of carbon credits. In particular, the costs for measuring and monitoring soil carbon content are expected to be high (Walcott *et al.*, 2009). More research is needed to determine the extent of transaction costs and the effect on the net economic benefit of carbon credits to the farmer. Future research should also investigate how costs and other barriers can be reduced so that small farmers can participate in global carbon markets.

6.7 CONCLUSIONS

The present study assesses the economic and environmental performance of biosaline (agro)forestry systems in three case studies in Bangladesh, India, and Pakistan. The results of these case studies indicate that (agro)forestry production systems on salt-affected soils can be economically viable in different settings. The NPV at a discount rate of 10% is 1.0 k€ ha^{-1} for the rice-tree agroforestry system on saline soils in coastal Bangladesh, 4.8 k€ ha^{-1} for the rice-wheat-tree agroforestry system on sodic/saline-sodic soils in Haryana, India, and 2.8 k€ ha^{-1} for the compact tree plantation on saline-sodic soils in Punjab province of Pakistan. Although the NPV is positive in all cases, the analyses indicate that the economic performance strongly depends on local conditions and the setup of the (agro)forestry system. Key factors that affect economic performance are the type and severity of salt-affectedness (which affect the type and setup of the agroforestry system and the tree species), the markets for wood products, the possibility of trading carbon credits, and the discount rate. Fuelwood and timber production costs in case studies 1 (Bangladesh) and 2 (India) as well as charcoal production costs in case study 2 (India) are below market prices and biomass from biosaline agroforestry systems is competitive with existing production chains. Production costs of fuelwood at a discount rate of 5% and of timber at discount rates of 5% and 10% are within the range of market prices in Pakistan but above this range for other discount rates, indicating the strong influence that the discount rate can have on the competitiveness of these products.

The analysis of GHG emissions from biosaline (agro)forestry shows that carbon sequestration occurs in all systems as a result of increased carbon stocks in belowground biomass and soil. Carbon sequestration amounts to 16 t CO_2-eq. ha^{-1} in case study 1 (Bangladesh), 26 t CO_2-eq. ha^{-1} in case study 2 (India), and 96 t CO_2-eq. ha^{-1} in case study 3 (Pakistan). This translates into economic values that increase the NPV by between 3 and 80% in case study 1, 1 and 14% in case study 2, and 9 and 129% in case study 3 depending strongly on the carbon credit price assumed in this study (1 – 15 € t^{-1} CO_2-eq.). The economic value of carbon sequestration by biosaline (agro)forestry depends on the discount rate, the carbon price, the amount of carbon sequestered, and, most importantly, on whether such systems are eligible for carbon trading. Eligibility requirements and other practical hurdles for small farmers wanting to participate in such trading schemes require further research. In addition to carbon sequestration, biosaline (agro)forestry systems provide opportunities for improving soil and water conditions and biodiversity levels. However, there are also environmental risks associated with biosaline (agro)forestry, particularly the potential invasiveness of salt-tolerant tree species and the impact on groundwater levels and quality. Both aspects require more systematic research to determine the conditions in which these negative impacts are likely and the management practices that can prevent such negative impacts.

The present study demonstrates that different biosaline (agro)forestry systems are economically viable. However, there are several constraints to the implementation and sustainability of biosaline (agro)forestry. Most importantly, biosaline (agro)forestry has high initial costs, a potential barrier for small farmers in developing countries such as Bangladesh, India, and Pakistan. Providing subsidies to cover establishment costs and low interest loans to farmers, as well as creating the possibility to take part in a carbon credit trading scheme or some other form of compensation for soil regeneration could improve the profitability and attractiveness of biosaline (agro)forestry for small farmers. Also important for the successful implementation of biosaline (agro)forestry systems is dissemination of knowledge about the various agroforestry systems and tree species required for different types and severity levels of salt-affected soils.

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6.9 APPENDIX 1: CASE STUDY DESCRIPTION

6.9.1 Case study 1: Coastal saline soils in Bangladesh

In Bangladesh, 1 Mha of land are salt-affected, mostly in the coastal zone of the Ganges - Brahmaputra River Delta. 29% of the salt-affected land is slightly affected, 30% moderately, 33% highly, and 8% extremely (Hossain, 2010). Sea water intrusion is the main cause of soil and water salinity in Bangladesh. Although sea water intrusion is a natural phenomenon at the coast, it is enhanced by overexploitation of freshwater resources and sea level rise. In coastal Bangladesh salinity is seasonal. During the dry season (especially the later part), water levels are low as a result of overexploitation and natural seasonal fluctuations in water levels. Sea water intrusion then reaches further inland, soil water evaporates, and salts build up in the soil. With the monsoonal rains, the (majority of the) salts are washed away again, rendering the soils non-saline or only slightly saline. In addition to the salinity problem, coastal areas in Bangladesh are often characterized by low soil fertility and are often affected by droughts in the dry season, when good quality irrigation water is not available or too expensive for small farmers (Haque, 2006). The seasonality of the salinity problem and low soil fertility in coastal Bangladesh allow the cultivation of rice only during the wet season. In non-salt-affected regions of Bangladesh, two or even three cropping seasons are possible.

The case study examines an agroforestry system that is constructed based on a combination of the currently most common agricultural land use in the coastal belt (rice production during the monsoon season) and results from existing coastal reforestation projects. Local Aman rice is transplanted during the monsoon (July and August) and harvested in November and December when soil salinity is low. A traditional, manual labour-intensive cultivation is typical in this region and assumed in this study. The rice fields are intersected by tree lines. Assuming a square field of one hectare, there are two lines, where each line has two rows of trees, and each row accounts for 50 trees. Thus, the tree spacing is 2 m by 2 m, the tree density is 200 trees per hectare, and the tree lines cover 4% of the land. *Acacia nilotica* is a common tree species in (agro)forestry systems. It produces wood as well as the by-products gum and seeds/pods, which are primarily used for medicinal purposes and fodder, respectively. For the purposes of the present study, gum is not included in the analysis because its collection is very labour intensive. A problem with *Acacia nilotica* in the coastal region of Bangladesh is that it is frequently affected by diseases. An alternative species is *Eucalyptus camaldulensis*, which is also a common species in coastal Bangladesh and which has higher biomass yields than *Acacia nilotica* in slightly and moderately saline soils. However, no by-products are considered for *Eucalyptus camaldulensis*. The establishment of the trees at the beginning of the monsoon season includes field levelling, ploughing, holing, fertilizer application, and planting of the six month-old seedlings. Maintenance in the following years includes canopy manipulation and pruning (annually for timber production), fertilizer application, and manual weeding in the first year of each rotation. This study considers fuelwood and timber as applications for biomass. Four rotations of five years each for fuelwood production and two rotations of ten years for timber are considered. A by-product of timber production is fuelwood from pruning and the branches at the final harvest.

6.9.2 Case study 2: Waterlogged, sodic/saline-sodic soils in Haryana, India

In India, approximately 7 Mha of land are salt-affected (Vashev *et al.*, 2010). Human-induced soil salinization and sodification in India has been primarily caused by irrigation. Irrigation-induced salinization and sodification of land occurs as a result of water seepage from irrigation canals causing groundwater tables to rise and dissolved salts to be moved to upper soil layers, irrigation water adding salts to the soil (Ritzema *et al.*, 2008), and/or the absence of natural drainage (Ram *et al.*, 2008). This case study focuses on waterlogged, salt-affected soils near the village Puthi in the Hisar district in the northern Indian state of Haryana. The soil properties indicate that the soil is slightly saline-sodic in the topsoil and moderately sodic in the subsoil (Ram *et al.*, 2008). Subsurface waterlogging exists throughout the year, and surface waterlogging during the monsoon season. Waterlogging in combination with salt-affected soils occurs as a result of water seepage from irrigation canals, brackish groundwater, and the absence of natural drainage (Ram *et al.*, 2008). The climate at the case study site is semi-arid monsoonal with intensely hot summers and cold winters (Ram *et al.*, 2008).

The agroforestry system investigated in this case study is a plantation that was established under the 'Biodrainage project for the reclamation of waterlogged areas' of the Haryana Forest Department (2008). The agroforestry system consists of rice cultivated in the monsoon season, wheat cultivated in the dry season, and tree lines intersecting the agricultural field. Trees are planted on ridges, and the ridges dissect the field from north to south. Each ridge is 1.3 m wide at the base and 0.6 m at the top, and the height of the ridges is 0.5 m. In total, 200 trees per hectare of the species *Eucalyptus tereticornis* are planted with a spacing of 1.5 m between trees. This results in approximately 4% of the land being attributed to the trees and 96% to agricultural production. Establishment of the trees includes the construction of ridges, preparing of slots (including digging holes and the application of fertilizers and gypsum), planting of approximately six month-old seedlings, and spot irrigation. Maintenance in the second year of the rotation includes spot irrigation (four times a year), weeding, hoeing, and fertilizer (farm yard manure and urea) application. No maintenance activities take place in subsequent years. The trees are harvested every five years, after which the tree stumps are allowed to coppice. Three rotations are assumed, allowing a project time of 15 years. The biomass produced is sold to wood processing contractors by auction. The wood is generally used for fuelwood, charcoal, and small timber, and these three applications are investigated in this study.

Case study data indicate that the groundwater table of the field is lowered by more than one-half meter compared to the adjacent fields with no trees and, in turn, agricultural crop yields are 30% higher than in the adjacent fields (Ram *et al.*, 2008). The biosaline (agro)forestry system of this case study is compared to conventional agriculture (without tree lines or other remediation) and to a conventional agricultural system with a subsurface and surface drainage system in order to test biosaline (agro)forestry's economic viability compared to alternative land use systems.

6.9.3 Case study 3: Saline-sodic soils in Indus river basin, Pakistan

In Pakistan, it is estimated that 3 to 6 Mha of land are salt-affected (Qureshi *et al.*, 2008; Vashev *et al.*, 2010). This is primarily irrigated agricultural land in the Indus River Basin in the Sind and Punjab provinces that became salt-affected as a result of improper water management (Qureshi *et al.*, 2008). The (agro)forestry system investigated in this case study is a forestry plantation on saline-sodic soils southwest of Faisalabad in the Pakistani province of Punjab. The soils of the case study site are extremely saline-sodic with electrical conductivity of the saturated soil extract (ECe) ranging from 15 to 54 dS m^{-1}, pH from 8.2 to 9.8, and sodium absorption ratio from 25 to 110. Punjab is characterized by a semi-arid climate, where summers are hot, winters are cool or cold, and the rainy season (July to September) brings the majority of the annual precipitation.

The system investigated in this study is a compact tree plantation. Intercropping is not considered because the case study site is so severely salt-affected that intercropping is not feasible. It is assumed that trees are planted at a spacing of 2 m by 3 m, which results in approximately 1,730 trees per hectare. The tree species planted is *Acacia nilotica*, which is a species commonly used in (biosaline) forestry plantations in Pakistan. The establishment and maintenance of the tree plantation include the following activities: field levelling, ploughing, layout of field, ditching, holing, application of gypsum and manure, and planting of saplings in field (saplings are raised in nursery and planted in fields at four to six months-old). The plants are irrigated weekly in the first month after establishment and by-weekly for the following six months. After that, trees are irrigated once a month for another three years. The only source of water for the case study plantation is brackish groundwater (electrical conductivity of 1.8 – 3.7 dS m^{-1}). The brackish groundwater is pumped with a diesel generator through a tube well, and the plants are spot irrigated with buckets. Other maintenance activities in the second year include manual weeding (annually) and restocking of plants that did not survive the first year. Thinning takes place in the fourth and sixth years when slow-growing and timber-unsuitable trees are harvested. This biomass is used for fuelwood. The remaining trees are harvested after ten years, with their stems used for timber and branches for fuelwood. By-products of *Acacia nilotica* are not collected because of the high tree density (and thorniness of *Acacia nilotica*) and the time-consuming collection.

6.10 APPENDIX 2: ADDITIONAL INPUT DATA

Table 6.4: Input data for the economic performance and GHG emission analysis

	Unit	Bangladesh	India	Pakistan
Unit costs				
Wheat	€ t^{-1}	-	148[n]	-
Rice	€ t^{-1}	144[g]	495[n]	-
Pods/seeds (*Acacia nilotica*)	€ kg^{-1}	0.02[b]	-	-
Tree seedlings	€ seedling^{-1}	0.03[e]	0.12[a]	0.04[o]
Urea	€ kg^{-1}	0.13[e]	0.11[d]	-
Triple super phosphate	€ kg^{-1}	0.24[e]	-	-
Mutriate of potash	€ kg^{-1}	0.27[e]	-	-
Manure/compost	€ t^{-1}	10.9[e]	5.9[d]	6.0[o]
Gypsum	€ t^{-1}	-	0.3[d]	27.1[o]
Machinery costs - tree harvest and extraction	€ (t dm)$^{-1}$	0.7[k]	0.7[k]	0.7[k]
Diesel	€ l^{-1}	0.5[g]	1.2[d]	0.9[o]
Wage	€ man-day^{-1}	1.1	1.9[a]	3.0[o]
Land rent	€ ha^{-1} y^{-1}	65[c]	-	-
Land selling value	€ ha^{-1}	-	1006[d]	724[o]
Application rates				
Urea	g tree^{-1}	40[e]	20[a]	-
Triple super phosphate	g tree^{-1}	30[e]	-	-
Mutriate of potash	g tree^{-1}	20[e]	-	-
Manure/compost	kg tree^{-1}	2[e]	5[d]	3[f]
Gypsum	kg tree^{-1}	-	3[d]	0.5[o]
Diesel requirements for chain saw	l h^{-1}	0.5[h]	0.5[h]	0.5[h]
Other diesel requirements (over lifetime of plantation)	l ha^{-1}	-	1525[k, j]	1060[o]
Labour requirements (total over plantations lifetime)				
Establishment	man-day ha^{-1}	20[g]	50[a]	19[o]
Irrigation	man-day ha^{-1}	-	10[a]	30[i]
Weeding, pruning, thinning	man-day ha^{-1}	22 (short rotation) 54 (long rotation)[g]	27[a]	39[o]
Pod/seed collection	man-day ha^{-1}	15[b]	-	-
Harvest and extraction	h (t dm)$^{-1}$	15[m]	9[a]	15[k]

a - Haryana Forest Department (2008);

b - Assuming the same price of *Acacia nilotica* pods/seeds and labour requirements for collecting them as in India (Viswanath *et al.*, 2001);

c - Department of Agricultural Extension (2005);

d - Estimates by Gurbachan Singh (Central Soil Salinity Research Institute);

e - Estimates by forest range officer in Khajura, Bangladesh;

f - Calculated based on data from Abdul Rasul Awan (Nuclear Institute for Agriculture and Biology, Pakistan) that half a trolley of farm yard manure is applied per acre where one trolley contains approximately 4 t of farm yard manure;

g - Estimate by Razzaque Akanda (Bangladesh Agricultural Research Institute) for coastal zone;

h - Riegelhaupt (2001);

i - Labour requirements are 4.3 h ha^{-1} irrigation^{-1}; irrigation is applied weekly in first month, by-weekly in second to sixth months, and monthly from the seventh month to 3.5 years;

j - Calculated based on machinery requirements for establishment of plantation (estimates by Gurbachan Singh, Central Soil Salinity Research Institute) and machinery specific diesel requirements (Smeets *et al.*, 2009c);

k - Trees are sold as standing stock to contractors who harvest the wood. For the calculation of biomass production costs, a manual labour-based harvesting system is assumed, applying labour requirements of 15 h (t dm)$^{-1}$ harvested and machinery costs of 0.7 € (t dm)$^{-1}$ ((FAO, 2008a); see also Table 5.5 in Chapter 5);

l – ILO (2009);

m - Based on FAO (2008a); see also Table 5.5 in Chapter 5.

n – HAU (2005);

o - Estimates by Abdul Rasul Awan (Nuclear Institute for Agriculture and Biology, Pakistan).

The fraction of the stem (which is suitable for timber) in the harvested biomass yield is 0.8 for *Eucalyptus tereticornis* and 0.46 for *Acacia nilotica* (Singh *et al.*, 2008). For *Eucalyptus camaldulensis* the same fraction as for *Eucalyptus tereticornis* is assumed. The remaining biomass is used for fuelwood or charcoal. Conversion efficiency of wood to charcoal is assumed to be 20% (based on interviews with wood traders near Puthi, India, April 2008). Conversion losses for timber production amount to 70 %. The transport and processing costs in all case studies are assumed to be the same because no case study-specific data are available. Transport costs are 0.1 € (t km)$^{-1}$, and transport distance is 15 km for charcoal and 90 km for timber. Fuelwood is assumed to be used locally, and the transport distance is zero. Processing costs amount to 8 € (t dm)$^{-1}$ for fuelwood, 16 € (t dm)$^{-1}$ for charcoal, and 51 € (t dm)$^{-1}$ for timber (Stille *et al.*, submitted). Values of energy content of tree biomass from *Acacia nilotica* of 20 GJ (t dm)$^{-1}$, *Eucalyptus camaldulensis* of 19 GJ (t dm)$^{-1}$, and *Eucalyptus tereticornis* of 20 GJ (t dm)$^{-1}$ are applied (Puri *et al.*, 1994; Brink, 2008). Market prices for fuelwood, charcoal, and timber for the three case studies are shown in Table 6.2.

In all cases a discount rate of 10% is applied. For case study 3 (Pakistan), the sensitivity to the discount rate is tested by also applying 5%, 15%, and 20% discount rates because of annual large fluctuations in Pakistan's interest rate (World Bank, 2010) and differences in the kind of loans farmers may get. Average currency exchange rates for 2007 are applied: 92 Bangladeshi Taka €$^{-1}$, 57 Indian Rupees €$^{-1}$, and 83 Pakistani Rupees €$^{-1}$ (OANDA, 2008).

Chapter 7

Summary and Conclusions

Current global energy supply is primarily based on fossil fuels and is widely considered to be unsustainable as a result of large greenhouse gas (GHG) emissions to the atmosphere, the finiteness of fossil fuels, the unequal geographic distribution of fossil fuels, and the failure of the system to provide billions of people with access to modern energy services. Bioenergy is considered an important option in making future global energy more sustainable. To begin with, bioenergy has a substantial growth potential and can therefore make a significant contribution to future energy supply. Secondly, if produced sustainably, bioenergy can reduce GHG emissions compared to fossil fuels. Thirdly, bioenergy is a versatile energy source that can be used to produce heat and electricity, as well as solid, liquid, and gaseous fuels. Fourthly, bioenergy resources are globally more evenly distributed than fossil fuels, decreasing the dependency of energy imports from a small number of countries and increasing local production of energy.

However, increasing global trade and consumption of bioenergy in industrialised countries has been accompanied by a growing concern about the environmental, ecological, and social impacts of (modern) bioenergy production. Many of these unintended and undesired effects of bioenergy production are linked to direct land use change (LUC) (conversion of one type of land to another) and indirect LUC (change in land use in one place induced by the expansion of bioenergy production in another place). However, producing bioenergy on degraded or marginal land may avoid these negative effects because this land is considered to be largely unsuitable and often economically unattractive for agricultural crop production. Perennial bioenergy production on degraded and marginal land can also sequester carbon, improve soil fertility, and reduce other soil degradation processes such as soil erosion, dispersion, and leaching as a result of above and belowground biomass growth. Moreover, perennial bioenergy crops cultivated on degraded and marginal land can increase the quantity and variability of biodiversity, especially if monoculture and large fields are avoided and a mixture of groundcover species are planted. In addition, producing bioenergy from degraded and marginal land can contribute to rural social and economic development by using land with little or no previous productivity.

Despite its potential advantages, the use of degraded and marginal land for bioenergy production has drawbacks that may limit its economic attractiveness and diminish its sustainability. The most important challenges are related to 1) difficult growing conditions that require much effort over a long period of time and that still often lead to lower productivity than high quality land, and 2) degraded land often being an important resource for poor rural communities, particularly those with no formal land rights. In addition, degraded and marginal land may still provide some ecosystem functions and support biodiversity levels similar to managed landscapes; maintaining or enhancing these values can also be a challenge. While these challenges are acknowledged in many studies, little is actually known about the implications for the technical and economic potential and the economic performance of bioenergy production on degraded and marginal land. More research is needed to determine the actual contribution degraded and marginal land can make to (bio)energy production and demand on different geographical scales (from local to global). Assessing the potential should focus on the extent and severity of degraded and marginal land, its availability and suitability for bioenergy production, and the yields. Moreover, the economic feasibility needs to be understood better, and more specific research on production costs and the economic potential of bioenergy production on different types and different severity levels of degraded and marginal land is necessary. In addition, there is a lack of knowledge about the environmental and social impacts of using this type of land for bioenergy production.

This thesis aimed to close some of these knowledge gaps. The main objective was to assess the technical and economic potentials, economic performance, and environmental impacts of bioenergy production on degraded and marginal land in different settings and at different geographical scales, ranging from local to global. To this end, the following research questions were addressed:

I What is the bioenergy production potential of degraded and marginal land in different settings and on different geographical scales?
II What is the economic performance of bioenergy production and its positive side effects in different settings of degraded and marginal land?
III What are the environmental impacts of bioenergy production in different settings of degraded and marginal land?

The research questions were addressed in Chapters 2 through 6. Each chapter evaluated bioenergy production on degraded and marginal land in different settings and geographical scales. Chapters 2 and 3 focused on palm oil production in Malaysia and Indonesia because of the recent global debate about the negative environmental impacts of palm oil. These chapters assessed the use of *Imperata* grasslands as an alternative to tropical rainforest or other land types. Chapter 4 evaluated cassava ethanol, jatropha oil, and fuelwood production from marginal semi-arid and arid land in eight sub-Saharan African countries because these regions have been under-researched and have substantially different conditions and requirements for bioenergy production than more humid regions. Chapters 5 and 6 investigated woody bioenergy production from salt-affected land focussing on a global scale (Chapter 5) and on local and national scales in Bangladesh, India, and Pakistan (Chapter 6). Salt-affected land is important to study because of its widespread occurrence globally and the difficulties it poses for agricultural production. Table 7.1 presents an overview of the settings, geographical scales, and the research questions that are addressed in these chapters. This is followed by a summary of each chapter and its main findings, and then the answers to the three research questions.

Table 7.1: Overview of the settings of bioenergy production on degraded and marginal land, geographical scales and research questions addressed in Chapters 2 through 6,

Chapter	**Settings**	**Geographical scale**	**Research question**		
			I	II	III
2	- Palm oil production systems on *Imperata* grasslands and other land types	- Local (case study in Northern Borneo, Malaysia)			•
3	- Land use patterns and palm oil production on *Imperata* grasslands	- National (Indonesia and Malaysia)	•		•
4	- Cassava ethanol, jatropha oil, and fuelwood production on marginal semi-arid and arid land	- Sub-continental to national (eight countries in sub-Saharan Africa)	•	•	
5	- Woody biomass production from forestry plantations on salt-affected soils	- Global to sub-continental (17 world regions)	•	•	
6	- Woody bioenergy production from agroforestry and forestry plantations on salt-affected soils	- Sub-continental to local (three case studies in South Asia)		•	•

Chapter 2 analysed the GHG emissions of crude palm oil (CPO) and palm fatty acid distillate (PFAD) production in northern Borneo (Malaysia), the transport of these products to the Netherlands, and their co-firing with natural gas for electricity production. In the case of CPO, conversion to biodiesel and the associated GHG emissions are also studied. The results demonstrated that land use change is the most decisive factor in overall GHG emissions and that palm oil energy chains based on land that was previously natural rainforest or peatland have such large emissions that they cannot meet the emission reduction target set by the European Commission's Renewable Energy Directive. However, if CPO production takes place on degraded land, management of CPO production is improved, or if the by-product PFAD is used for electricity production, emission reduction criteria can be met, and palm oil-based electricity can be considered sustainable from a GHG emission point of view. Even though the biodiesel base case on logged-over forest can reduce emissions by 30%, other cases, such as oil palm plantations on degraded land and improved management, can achieve emissions reductions of more than 150%. This would turn oil palm plantations into carbon sinks. Given these considerations, this chapter concluded that in order for bio-electricity and biodiesel to be sustainably produced from

palm oil and its derivatives, degraded land should be used for palm oil production and plantation management should be improved.

Based on the importance of LUC in the GHG balance of palm oil based energy chains, **Chapter 3** compiled and analysed national level data on LUC and its causes in Indonesia and Malaysia over the past 30 years. This chapter also explored the role that palm oil has played in past LUC and that projected growth in palm oil production may play in LUC until 2020 and suggested strategies to minimize negative effects. Data collection for the study revealed that the quality and quantity of data on LUC on a national scale over time are limited. Despite these uncertainties, the overview of past LUC indicated that large changes in land use have occurred in Indonesia and Malaysia. In Indonesia, LUC can primarily be characterized by forest cover loss on 40 million hectare (Mha) of land, a 30% reduction in forest land. Deforestation in Malaysia has been smaller in both absolute and relative terms, with a forest cover loss of nearly 5 Mha, a 20% reduction in forest land. Other large changes in Malaysia occurred in permanent cropland (excluding oil palm), which has decreased rapidly since the early 1990s, and in land under oil palm cultivation, which experienced a sharp increase. Projections of additional land demand for palm oil production in 2020 ranged from 1-28 Mha in Indonesia. This demand can be met to a large extent by degraded land if no further deforestation is assumed. In Malaysia, expansion projections ranged from 0.06 to five Mha, but only the lowest projection of oil palm expansion is feasible if only degraded land may be used. The role of palm oil production in future LUC depends on the size of the projected expansion as well as agricultural management factors such as earlier replanting with higher yielding plants and establishment of new plantations on degraded land. The current use of degraded land needs to be investigated in order to reduce possible indirect LUC, land tenure conflicts, and other social impacts. In addition to minimizing direct and indirect LUC by the palm oil sector, measures that reduce deforestation triggered by other causes must also be implemented. A key element for doing so is better planning and governance of land use, which entails more appropriate demarcation of forest land and protection of land that still has forest cover, improved monitoring of land use, and more research to uncover the complexities and dynamics of the causes and drivers of LUC.

Chapter 4 assessed the current technical and economic potential of three bioenergy production systems (cassava ethanol, jatropha oil, and fuelwood) in semi-arid and arid regions of eight sub-Saharan African countries: Botswana, Burkina Faso, Kenya, Mali, Senegal, South Africa, Tanzania, and Zambia. The results indicated that the availability of land for energy production ranges from 2% (1.3 Mha) of the total semi-arid and arid area in South Africa to 21% (12 Mha) in Botswana. Land availability for bioenergy production is restricted mainly by agricultural land use, but also by steep slopes and biodiversity protection. The current total technical potential for the semi-arid and arid regions of the eight countries was calculated to be approximately 300 PJ y^{-1} for cassava ethanol production, 600 PJ y^{-1} for jatropha oil, and 4,000 PJ y^{-1} for fuelwood. The analysis of economic potentials showed that in many semi-arid regions, cassava ethanol, jatropha oil, and fuelwood can compete economically with the reference energy sources (gasoline, diesel, and fuelwood). However, in most arid regions of sub-Saharan Africa production costs of fuelwood, jatropha oil, and cassava ethanol are often above average national market prices for fuelwood, diesel, and gasoline. Despite high production costs in some regions, it is important to investigate and invest in sustainable bioenergy production in semi-arid and arid regions of sub-Saharan Africa because of its potential to contribute to and drive rural economic and social development.

Chapter 5 assessed the extent and location of salt-affected soils worldwide and their current land use/cover as well as the current technical and economic potential of biomass production from forestry plantations on these soils (biosaline forestry). The global extent of salt-affected land amounts to approximately 1.1 x 10^9 hectare (Gha), of which 14% is classified as forest, wetlands, or (inter)nationally protected areas and is considered unavailable for biomass production because of sustainability concerns. For the remaining salt-affected area (1.0 Gha), this study found an average biomass yield of 3.1 t dry matter (dm) ha^{-1} y^{-1} (ranging from 0 to 27 t dm ha^{-1} y^{-1}) and a global technical potential of 56 EJ y^{-1} (equivalent to 11% of current global primary energy consumption). If land currently used for agricultural production is also considered unavailable because of sustainability concerns, the technical potential decreases to 44 EJ y^{-1}. The global economic potential of biosaline forestry at production costs of 2 € GJ^{-1} or less was calculated to be 21 EJ y^{-1} when including agricultural land and 12 EJ y^{-1} when excluding agricultural land. At production costs of up to 5 € GJ^{-1}, the global economic potential increases to 53 EJ y^{-1} when including agricultural land and to 39 EJ y^{-1} when excluding agricultural land. Biosaline forestry may contribute even more significantly to energy supply in certain regions, *e.g.* Africa. Biosaline forestry has numerous additional benefits that make it interesting to investigate and invest in, including its potential to improve soil, generate income from previously low-productive or unproductive land, and sequester carbon.

Chapter 6 assessed the economic performance (*i.e.* net present value (NPV) and production costs) and environmental performance (primarily the impact on greenhouse gas emissions) of different biosaline (agro)forestry systems in three cases in South Asia. The economic impact of trading carbon credits generated by biosaline (agro)forestry was also assessed as a potential additional source of income. The NPV at a discount rate of 10% is 1.0 k€ ha^{-1} for a rice-tree agroforestry system on saline soils in coastal Bangladesh (case study 1); 4.8 k€ ha^{-1} for a rice-wheat-tree agroforestry system on sodic/saline-sodic soils in Haryana, India (case study 2); and 2.8 k€ ha^{-1} for a compact tree plantation on saline-sodic soils in Punjab province of Pakistan (case study 3). The GHG balance of the three systems shows carbon sequestration rates of 16 t CO_2-eq. ha^{-1} in Bangladesh, 26 t CO_2-eq. ha^{-1} in India, and

96 t CO_2-eq. ha^{-1} in Pakistan. This translates into economic values that increase the NPV by 3-80% in case study 1, 1-14% in case study 2, and 9-129% in case study 3, depending on the carbon credit price assumed in this study (1-15 € t CO_2-eq.). Although the NPV is positive in all three cases, the analysis indicated that the economic performance strongly depends on the type and severity of salt-affectedness (which affect the type and setup of the agroforestry system, the tree species, and the yield), tree rotation length, the markets for wood products, the possibility of trading carbon credits, and the discount rate. A simple extrapolation of the results suggests a technical bioenergy production potential from biosaline (agro)forestry of 1.2 EJ y^{-1} (approximately 3% of the total current primary energy consumption of the three countries) and a climate change mitigation potential of 43 Mt CO_2-eq. that could be sequestered annually (approximately 3% of the three countries' GHG emissions related to energy consumption).

Based on the findings of Chapters 2-6, the following answers to the research questions and recommendations for future research are given.

I What is the bioenergy production potential of degraded and marginal land in different settings and on different geographical scales?

The technical bioenergy production potential of degraded and marginal land was assessed in this thesis in different settings and geographical scales. Table 7.2 gives an overview of the potentials found. The results of this thesis indicate a technical potential of approximately 56 EJ y^{-1} for salt-affected land (Table 7.2). Combining this potential found in this thesis with the 32 EJ y^{-1} bioenergy production potential of human-induced degraded land (excluding salt-affected land) estimated by Nijsen *et al.* (submitted) (see also Section 1.2.1) indicates that the technical bioenergy production potential of naturally and human-induced degraded and marginal land amounts to approximately 90 EJ y^{-1}. This is equivalent to approximately 18% of the approximately 514 EJ y^{-1} current global primary energy consumption (OECD/IEA, 2010c). A comparison with results found in the literature indicates that this potential is in-between the estimate of Dornburg *et al.* (2010; 70 EJ y^{-1}) and the upper end of the range given by Hoogwijk *et al.* (2003; 8-110 EJ y^{-1}) and Schubert *et al.* (2009; 34-120 EJ y^{-1}). Accounting for the potential bioenergy production on non-salt-affected arid and semi-arid regions in sub-Saharan Africa and elsewhere could further increase this potential.

However, there are also factors that are accounted for only in a limited manner in the assessment of the bioenergy production potential of degraded and marginal land. The two most important factors are the possible exacerbation of water shortages in water-scarce regions and the current use and function of degraded and marginal land. Both factors could reduce the potential of sustainable bioenergy production, but the magnitude to which they would do so is currently unclear.

Table 7.2: Overview of the technical bioenergy potential of degraded and marginal land in different settings and geographical scales, taking constraints related to sustainable land use into account

Chapter	Settings	Geographical scale	Extent		Average yield		Potential	
			Mha		$t\ ha^{-1}\ y^{-1}$ [a]		$EJ\ y^{-1}$	
3	Palm oil production on *Imperata* grasslands	National: Malaysia	1		4.3	6.1[b]	0.1	0.2[b]
		Indonesia	12		3.5	5.9[b]	1.1	1.9[b]
4	Cassava ethanol, jatropha oil, and fuelwood production on marginal semi-arid and arid land	National: Botswana	8	4[c]	5.5	0.7[c]	0.8	0.1[c]
		Burkina Faso	2	0	10.0	6.3	0.3	1.0
		Kenya	3	3	12.4	8.9	0.7	0.5
		Mali	3	6	8.1	2.7	0.5	0.5
		Senegal	1	1	7.4	5.7	0.1	0.1
		South Africa	1	0	8.7	1.1	0.1	0
		Tanzania	2	0	12.4	n.a.	0.5	0
		Zambia	2	0	9.5	n.a.	0.3	0
		Sub-continental: Eight sub-Saharan countries combined	22	13	7.2	4.1	3.2	1.1
5	Woody biomass production from forestry plantations on salt-affected soils	Global	971		3.1		56.2[d]	
		Regions with most potential: Oceania	144		7.6		20.2	
		former USSR	117		4.7		10.0	
		South America	57		5.2		5.4	
6	Woody biomass production from (agro)forestry plantations on salt-affected soils	Regional: Bangladesh, India and Pakistan combined	11		6.0		1.2	

a – Average yield given for palm oil in tonnes of crude palm oil (CPO) $ha^{-1}\ y^{-1}$ and for fuelwood and woody biomass in tonnes of dry matter (t dm) $ha^{-1}\ y^{-1}$.

b – Yields in 2020 for base case and improved case.

c – The extent, average yield, and potential are shown for semi-arid (left column) and arid (right column) regions for fuelwood only. Yields and potentials of cassava ethanol and jatropha oil are listed in Table 4.1 and Table 4.4 (Chapter 4).

d – The extent, yield, and potential of woody biomass production on salt-affected land are shown for the areal extent with agricultural land. The results for excluding agricultural land because of sustainability concerns are shown in Chapter 5.

n.a. – not applicable

Regarding possible exacerbation of water shortages, more research is needed on the hydrological impacts of (large-scale) bioenergy production at different geographical scales;

on the economic, social, and environmental impacts of increased competition for water; on the effects of perennial bioenergy crops on water infiltration and soil moisture retention; and on how these aspects affect the sustainable potential of bioenergy production on degraded and marginal land. The basis for any such assessment is spatially-explicit datasets on groundwater depth and quality. However, particularly on a global and continental scale, such datasets do not exist as found when assessing the technical and economic potential of bioenergy production on salt-affected land (see Chapter 5). Research efforts should therefore first focus on creating more national level groundwater data so that also detailed continental and global level datasets can be created.

Regarding the current use and function of degraded and marginal land, this thesis has attempted to exclude land based on the main competing uses and functions, *i.e.*, food and feed production and high and unique biodiversity areas. However, the datasets applied cannot account for all uses. An example of land uses and functions that are particularly difficult to accurately capture are the use of land for livestock grazing, hunting, and gathering as well as the ecosystem and cultural services provided by degraded and marginal land. Not accounting for these uses and functions may result in bioenergy production displacing these uses and functions or in potentials being too high. Therefore, future research needs to assess these uses and services and investigate how they affect land availability for bioenergy production. In addition, when planning or investigating actual projects, an assessment of current land use, ownership, and functions needs to be conducted in order to avoid or at least minimize unsustainable land use, loss of ecosystem functions, and negative social and environmental consequences of bioenergy production.

II What is the economic performance of bioenergy production and its positive side effects in different settings of degraded and marginal land?

The results of this thesis show that biomass and bioenergy production on degraded and marginal land may be economically feasible in various regions and may contribute to local and regional biomass and/or energy needs. Chapter 6 analyzed the economic performance of (agro)forestry systems on different types of salt-affected soils in South Asia in terms of net present value (NPV). Although the NPV is positive in the three cases investigated, the analyses indicate that economic performance strongly depends on the type and severity of salt-affectedness (which affect the type and setup of the (agro)forestry system and the tree species), tree rotation length, the markets for wood products, the possibility of trading carbon credits, and the discount rate. Incorporating the economic value of carbon sequestration by the (agro)forestry systems may increase the NPV by up to 129% depending on the carbon credit price assumed in the analysis (1-15 € t^{-1} CO_2-eq.) and on whether the (agro)forestry system is eligible for carbon trading. Eligibility requirements and practical hurdles for small farmers wanting to participate in carbon trading schemes require further attention.

The bioenergy costs of different production systems on degraded and marginal land were found to be competitive with relevant market prices. Examples of these systems are fuelwood production in many semi-arid regions in sub-Saharan Africa, jatropha oil and cassava ethanol production in a few semi-arid regions in sub-Saharan Africa (Chapter 4), and fuelwood and charcoal production in (agro)forestry plantations on salt-affected soils in South Asia (Chapter 6). However, fuelwood, jatropha oil, and cassava ethanol production costs in most arid regions of sub-Saharan Africa are often above average national market prices of gasoline, diesel, and fuelwood. Despite high production costs, sustainable bioenergy production in these regions may still be desirable because it can be a potential driver of rural economic and social development. Potential developmental benefits include local production and supply of energy in rural regions, creating new markets for agricultural products, and helping to generate more income for rural populations.

Based on the production costs, cost-supply curves were constructed to determine the economic potential of bioenergy production on degraded and marginal land (see Figure 4.4 and Figure 5.4). This thesis showed that the economic potential can be significant for many regions. For example, Chapter 4 estimated the economic potential of fuelwood production in semi-arid Tanzania to be 103 PJ y^{-1} at production costs of up to 2 € GJ^{-1}. This is equivalent to 16% of Tanzania's current primary energy consumption. Another example was given in Chapter 5, where bioenergy production from salt-affected soils is shown to potentially contribute to energy supply, particularly in Africa. The economic potential at production costs of 2 € GJ^{-1} or less was calculated to be 8 EJ y^{-1}, which is approximately 28% of the current total primary energy consumption in Africa. The global economic potential of biomass production from salt-affected soils (when including agricultural land) was determined to be 21 EJ y^{-1} (or 4% of global primary energy consumption) at production costs of 2 € GJ^{-1} or less. The economic potential increases to 53 EJ y^{-1} when biomass produced at costs of 5 € GJ^{-1} or less are included. If agricultural land is excluded for sustainability reasons, the economic potential of biosaline forestry decreases to 12 EJ y^{-1} at production costs of 2 € GJ^{-1} or less and to 39 EJ y^{-1} at production costs of 5 € GJ^{-1} or less.

III What are the environmental impacts of bioenergy production in different settings of degraded and marginal land?

This thesis found that woody bioenergy production on degraded and marginal land can have both positive and negative environmental impacts. Potential positive impacts are carbon sequestration, an increase in soil fertility, improved water infiltration and soil moisture retention, a reduction in soil degradation processes, and amelioration of soil salinity and sodicity. Detailed assessments of the GHG balance for two examples of bioenergy production from degraded and marginal land - palm oil-based energy production from *Imperata* grassland in Malaysia (Chapter 2) and biomass production in (agro)forestry systems on salt-affected soils in South Asia (Chapter 6) - indicated that bioenergy from degraded and marginal land can achieve emissions reductions of more than 100% compared to fossil fuels and turning bioenergy plantations on degraded and marginal land into carbon sinks. Chapter 2 further revealed that land use change is the most decisive factor in overall GHG emissions of palm oil energy production and that, if previously natural rainforest or peatland are used instead of degraded land, the emission reduction targets set by the European Commission's Renewable Energy Directive cannot be met. However, if palm oil production takes place on degraded land and plantation management is improved, emission reduction criteria can be met, and palm oil-based electricity can be considered sustainable from a GHG emission point of view.

Despite these possible positive impacts, bioenergy production on degraded and marginal land may also pose environmental risks to soils, biodiversity, and water. For example, the assessment of (agro)forestry systems on salt-affected soils in South Asia (Chapter 6) indicated that there is the risk that tree species tolerant to the more difficult growing conditions of degraded and marginal land may become invasive and weedy even in non-degraded/marginal areas. It is therefore suggested that mainly native species are planted and species specific management is applied in order to minimize this risk. Another risk is the possible exacerbation of water shortages in already water-scarce regions. As indicated in the answer to research question 1, more research is needed on the economic, social, and environmental (particularly hydrological) impacts of increased competition for water by (large-scale) bioenergy production on degraded and marginal land. In addition, more systematic research is needed to determine the conditions (*e.g.* type and severity of degradation) under which bioenergy can have negative impacts on biodiversity so that these can be avoided or at least minimized. Moreover, Chapter 4 stressed that collecting detailed national-level datasets on biodiversity is also important in order to be able to better account for biodiversity in the assessments of bioenergy potentials.

In addition, the social impacts and socio-economic performance of bioenergy production on degraded and marginal land must be investigated in order to better understand its overall performance and sustainability. Two important aspects of such an assessment are 1) the potential (in)direct land use change induced by the use of degraded land for bioenergy production and the associated opportunities (related to, for example, income generation from previously low-productive land and the regeneration of soils) and risks (related to, for example, the displacement of hunter/gathers and nomadic pastoralist communities), and 2) the conditions under which these risks can be minimized.

In summary, three main conclusions can be drawn. Firstly, bioenergy production on degraded and marginal land has a substantial potential for contributing to global energy consumption (approximately 18% of current global primary energy consumption). Secondly, the analysis of economic performance suggests that bioenergy production on these land types can in many cases be competitive with other bioenergy sources and even fossil fuels. Thirdly, perennial bioenergy production on degraded and marginal land can have both positive (*e.g.* carbon sequestration, improvements in soil fertility, and reduction in soil degradation) and negative (*e.g.* exacerbation of water shortages in water-scarce regions) environmental impacts; more research is needed to determine the conditions under which bioenergy production results in negative environmental impacts so that these can be avoided or at least minimized. Given these findings, the following key recommendations are identified:

1) Demonstration projects in different settings should be implemented in order to gain experience in establishing and operating sustainable bioenergy production on degraded and marginal land.
2) Infrastructure development and capacity building are needed. With regard to capacity building, this is particularly necessary in terms of the setup and management of sustainable bioenergy production on different types and severity levels of degraded and marginal land and for minimizing the possible environmental risks.
3) The potential positive side effects of bioenergy production - such as the possibilities to restore degraded land, generate income from previously low-productive or unproductive land, sequester carbon, improve water retention, and control erosion - are important additional reasons for investigating and investing in bioenergy production on degraded and marginal land. Incorporating the economic value of these positive side effects in economic assessments can make bioenergy production on degraded and marginal land more attractive. Incentives and policies are needed for internalizing these externalities.
4) Although bioenergy production on degraded and marginal land can have many positive environmental impacts, it must still comply with sustainability criteria. This requires certification, just like bioenergy production on any other type of land.

Hoofdstuk 7

Samenvatting en Conclusies

De huidige mondiale energievoorziening is hoofdzakelijk gebaseerd op fossiele brandstoffen. Dit wordt algemeen beschouwd als niet duurzaam vanwege de grote uitstoot van broeikasgassen, de eindigheid van fossiele brandstoffen, en hun ongelijke geografische spreiding. Daarnaast is er het falen van het energievoorzieningssysteem om in ontwikkelingslanden een paar miljard mensen toegang tot moderne energie te geven. Bio-energie wordt beschouwd als een belangrijke optie om een duurzamere mondiale energievoorziening te realiseren. Ten eerste heeft bio-energie een aanzienlijk (groei)potentieel en kan het een belangrijke bijdrage leveren aan de toekomstige energievoorziening. Ten tweede kan bio-energie, mits duurzaam geproduceerd, de uitstoot van broeikasgassen in vergelijking met fossiele brandstoffen verminderen. Ten derde is bio-energie een veelzijdige energiebron, die kan worden gebruikt voor de productie van zowel warmte en elektriciteit, als vaste, vloeibare en gasvormige brandstoffen. Ten vierde zijn de potentiële gebieden waar biomassa als grondstof voor de energievoorziening gewonnen kan worden wereldwijd gelijkmatiger verdeeld dan fossiele brandstoffen. Daarom kan het gebruik hiervan tot een vermindering van de afhankelijkheid van energie-import uit een klein aantal landen leiden en tot een toename van lokale productie en consumptie van energiedragers.

De toenemende wereldwijde handel in bio-energie en de consumptie ervan in geïndustrialiseerde landen gaat echter gepaard met een groeiende bezorgdheid over de sociale, ecologische en milieu-effecten van (moderne) bio-energie productie. Veel van deze onbedoelde en ongewenste effecten van bio-energie productie staan in verband met directe veranderingen in landgebruik (land use change, LUC) - conversie van het ene type landgebruik naar het andere - en indirecte LUC- verandering van het landgebruik op een plaats die veroorzaakt wordt door uitbreiding van de bio-energieproductie op een andere plaats). De productie van bio-energie op gedegradeerd en marginaal land kan deze negatieve effecten in hoge mate voorkomen, omdat dergelijk land thans wordt beschouwd als grotendeels ongeschikt en vaak niet aantrekkelijk voor landbouw. Productie van meerjarige bio-energiegewassen op gedegradeerde en marginale gronden kan als gevolg van een toename van boven- en ondergrondse groei van biomassa ook leiden tot opslag van koolstof in de bodem, de bodemvruchtbaarheid verbeteren en bodemaantasting zoals erosie, dispersie, en uitloging verminderen. Bovendien kunnen meerjarige bio-energiegewassen op deze gronden de hoeveelheid en variabiliteit van de biodiversiteit verhogen, vooral als monoculturen en grote plantages worden vermeden en een combinatie van verschillende bodembedekkers worden geplant. Verder kan de productie van bio-energie op deze gronden bijdragen aan de sociale en economische ontwikkeling van rurale gebieden.

Tegenover deze mogelijke voordelen staan nadelen die de economische mogelijkheden en duurzaamheid kunnen verminderen of ondermijnen. Belangrijke uitdagingen zijn: 1) de moeilijke groeiomstandigheden die veel inspanning over een lange termijn vergen om tot productie te komen maar die toch kunnen leiden tot een lagere productiviteit in vergelijking tot land met een hoge grondkwaliteit, en 2) het feit dat gedegradeerde en marginale gronden vaak een belangrijke resource zijn voor lokale gemeenschappen, in het bijzonder diegene zonder formele landrechten. Daarnaast kunnen ook gedegradeerde en marginale gronden verschillende functies in het ecosysteem hebben en een biodiversiteitniveau hebben dat vergelijkbaar is met die van beheerde landschappen. De instandhouding of versterking van deze waarden kunnen beperkingen opleveren. Hoewel deze uitdagingen zijn onderkend in vele studies is er eigenlijk weinig bekend over de gevolgen ervan voor de technische en economische potenties van winning van bio-energie in deze gebieden. Er bestaat met name een gebrek aan kennis over de condities waaronder milieu- en sociale risico's ontstaan en hoe deze risico’s kunnen worden vermeden. Daarnaast is meer onderzoek nodig om te kunnen bepalen welke bijdrage gedegradeerde en marginale gronden in de praktijk kunnen leveren aan het voorzien in de vraag naar (bio)energie op mondiale, regionale, nationale en lokale schaal.

Een analyse van de potenties van energieteelt op gedegradeerde en marginale gronden dient zich te richten op de omvang en de staat van deze gronden, de beschikbaarheid en geschiktheid van deze gronden voor bio-energieproductie, en de opbrengsten die gehaald kunnen worden. Bovendien zijn de kosten van de productie van biomassa op deze gronden belangrijk om vast te kunnen stellen of bio-energieproductie alhier economisch haalbaar is. Daarvoor is specifiek onderzoek noodzakelijk.

Dit proefschrift is gericht op het invullen van een aantal van deze kennishiaten. Daartoe zijn de volgende onderzoeksvragen geformuleerd:

I Wat is het potentieel van bio-energieproductie op gedegradeerde en marginale gronden binnen verschillende contexten en op verschillende geografische schaalniveaus?

II Hoe ziet de economie van bio-energieproductie op deze gronden eruit en wat zijn mogelijk bijkomende voordelen?

III Wat zijn milieu-effecten van bio-energieproductie op gedegradeerde en marginaal gronden en hoe hangen deze af van de context waarbinnen de energieproductie gebeurt?

De onderzoeksvragen worden behandeld in de hoofdstukken 2 tot en met 6. Elk hoofdstuk evalueert bio-energieproductie op gedegradeerde en marginale gronden binnen verschillende contexten en op verschillende geografische schaalniveaus.

Hoofdstukken 2 en 3 richten zich op de productie van palmolie in Maleisië en Indonesië, aansluitend bij het recente wereldwijde debat over de negatieve milieu-effecten van palmolie productie in deze landen. De hoofdstukken gaan in op het gebruik van *Imperata* graslanden als een alternatief voor het onttrekken van grond aan tropisch regenwoud of andere soorten land.

Hoofdstuk 4 evalueert de productie van cassave-ethanol, jatropha-olie, en brandhout op marginale semi-aride en aride gronden in acht landen in Sub-Sahara Afrika. Hiervoor is gekozen omdat deze regio in het verkennen van de mogelijkheden van bio-energie onderbelicht is gebleven en dit gebied wezenlijk verschillende voorwaarden en eisen voor de productie van bio-energie heeft vergeleken met vochtiger gebieden.

Hoofdstukken 5 en 6 zijn gericht op bio-energieproductie uit houtige gewassen op verzilt land op een mondiale schaal (hoofdstuk 5) en op lokale en nationale schaal, toegespitst op Bangladesh, India en Pakistan (hoofdstuk 6). Aandacht voor de potenties die verzilt land biedt is van belang vanwege de grote omvang van het areaal op aarde. Tegelijk zijn er de moeilijkheden die verzilt land oplevert voor landbouw.

Tabel 7.1 geeft een overzicht van de context, de geografische schaalniveaus, en de onderzoeksvragen die worden behandeld in deze hoofdstukken. In dit hoofdstuk wordt hiervan een samenvatting gegeven en worden de belangrijkste conclusies gepresenteerd. Tevens wordt een antwoord gegeven op de drie onderzoeksvragen.

Tabel 7.1: Overzicht van de verschillende contexten van bio-energieproductie op gedegradeerde en marginale gronden en de geografische schaalniveaus die in de hoofdstukken 2 tot en met 6 worden gesproken in relatie tot de drie onderzoeksvragen.

Hoofdstuk	**Contexten**	**Geografisch schaalniveau**	**Onderzoeksvraag**		
			I	II	III
2	- Palmolie productie op *Imperata* graslanden en andere typen land	- Lokaal (case studie in Noord-Borneo, Maleisië)			•
3	- LUC en palmolieproductie op *Imperata* graslanden	- Nationaal (Indonesië en Maleisië)	•		•
4	- Cassave–ethanol, jatropha-olie en brandhout productie op marginale semi-aride en aride gronden	- Subcontinentaal - nationaal (acht landen in Sub-Sahara Afrika)	•	•	
5	- Houtige biomassaproductie via bosbouw op verzilte bodems	- Mondiaal - subcontinentaal (17 wereld regio's)	•	•	
6	- Houtige biomassaproductie via land- / bosbouw op verzilte bodems	- Subcontinentaal - lokaal (drie case studies in Zuid Azië)		•	•

Hoofdstuk 2 analyseert de uitstoot van broeikasgassen bij de productie van ruwe palmolie (crude palm oil, CPO) en palm vetzuur destillaat (palm fatty acid distillate, PFAD) in het noorden van Borneo (Maleisië), het vervoer van deze producten naar Nederland, en hun

co-verbranding in Nederland met aardgas voor de productie van elektriciteit. In het geval van CPO is ook de uitstoot van broeikasgassen bij de conversie naar biodiesel bestudeerd. De resultaten tonen aan dat veranderingen in het landgebruik de doorslaggevende factor in de totale uitstoot van broeikasgassen zijn. Palmolieproductie op land dat voorheen bestond uit natuurlijk regenwoud of veengronden, resulteert in een dermate grote uitstoot van broeikasgassen dat de emissiereductie doelstelling zoals geformuleerd in de Renewable Energy Directive van de Europese Commissie (EC-RED), niet kan worden gehaald. Echter, als de productie van palmolie plaatsvindt op gedegradeerde gronden, het plantage-beheer is verbeterd, of als het bijproduct PFAD (gebaseerd op palm olie geproduceerd op land dat voorheen logged-over forest was) wordt gebruikt voor de productie van elektriciteit, kan wel worden voldaan aan de emissiereductie criteria. Dan kan elektriciteitsopwekking met palmolie vanuit het oogpunt van de broeikasgasbalans gezien wel als duurzaam worden beschouwd. Hoewel de biodiesel base case (logged-over forest) de uitstoot kan verminderen met 30% in vergelijking met diesel uit fossiele energiebronnen, kunnen andere gevallen, zoals de aanleg van palmolieplantages op gedegradeerd land en een beter beheer, emissiereducties van meer dan 150% bereiken. Dit zou palmolieplantages gecombineerd met netto koolstofopslag mogelijk maken. Dit hoofdstuk concludeert derhalve dat, om bio-elektriciteit en biodiesel uit palmolie en zijn derivaten op duurzame wijze te produceren, gedegradeerd land moet worden gebruikt en plantage-beheer moet worden verbeterd.

Gegeven het effect van veranderingen in landgebruik (LUC) op de broeikasgasbalans van palmolie energieketens, worden in **hoofdstuk 3** gegevens over LUC en de oorzaken ervan in Indonesië en Maleisië verzameld en geanalyseerd voor de afgelopen 30 jaar. In dit hoofdstuk wordt ook onderzocht welke rol palmolieproductie heeft gespeeld in historische LUC en welke rol de verwachte groei in palmolieproductie zou kunnen spelen tot 2020. Verder worden in dit hoofdstuk strategieën beschreven om negatieve effecten van palmolieproductie te minimaliseren. Uit het onderzoek blijkt dat de kwaliteit en beschikbaarheid van historische gegevens over LUC in Maleisië en Indonesië te wensen overlaat. Ondanks de onzekerheden geeft het overzicht van de historische LUC aan dat grote veranderingen in landgebruik hebben plaatsgevonden in Indonesië en Maleisië. In Indonesië kan LUC vooral gekenmerkt worden door een verlies aan bosareaal van 40 miljoen hectare (Mha), gelijk aan een reductie van 30% van het nationale bosareaal. Ontbossing in Maleisië is kleiner van schaal, zowel in absolute als in relatieve zin, met een bosareaalverlies van bijna 5 Mha, hetgeen gelijk staat aan een vermindering van het areaal met 20%. Andere grote veranderingen in Maleisië zijn een snel gedaalde hoeveelheid permanent akkerland (met uitzondering van land voor palmolieproductie) sinds de vroege jaren 1990 en een sterke stijging van het landgebruik voor oliepalmteelt. Projecties voor de extra ruimtevraag voor de productie van palmolie in 2020 variëren voor Indonesië van 1 tot 28 Mha. Aan deze vraag kan voor een groot deel worden voldaan door uitbreiding van oliepalmteelt op gedegradeerde grond, zonder verdere ontbossing. De projecties voor Maleisië betreffen uitbreiding van 0,06 tot 5,0 Mha, maar alleen de kleinste uitbreiding is haalbaar wanneer enkel gedegradeerde gronden mogen worden gebruikt. De rol van palmolieproductie in toekomstige LUC is afhankelijk van de omvang van de uitbreiding en van managementfactoren zoals het eerder herplanten van oliepalmen met een hogere opbrengst en het realiseren van nieuwe aanplant op gedegradeerd land. Het huidige gebruik van gedegradeerde gronden dient te worden onderzocht om mogelijke indirecte LUC, grondbezit conflicten, en andere sociale gevolgen te kunnen beperken. In aanvulling op het minimaliseren van de directe en indirecte LUC door de palmoliesector, moeten ook maatregelen worden genomen om ontbossing - veroorzaakt door andere factoren – te verminderen. Een belangrijk element hierbij is een betere planning en beheer van landgebruik. Dit verlangt een betere afbakening van bosgebieden, bescherming van bebost land, een betere monitoring van het landgebruik en meer onderzoek om de complexiteit en dynamiek van de oorzaken en drijfveren van LUC te begrijpen.

Hoofdstuk 4 evalueert het huidige technische en economische potentieel van drie bio-energie productiesystemen (cassave-ethanol, jatropha-olie, en brandhout) in semi-aride en aride gebieden van acht landen in Sub-Sahara Afrika: Botswana, Burkina Faso, Kenia, Mali, Senegal, Zuid-Afrika, Tanzania en Zambia. De resultaten geven aan dat de beschikbaarheid van grond voor de teelt van energiegewassen varieert van 2% (1,3 Mha) van het totale areaal aan semi-aride en aride gebied in Zuid-Afrika tot 21% (12 Mha) in Botswana. De beschikbaarheid wordt voornamelijk beperkt door gebruik van land voor landbouw, maar ook door steile hellingen en bescherming van de biodiversiteit. Het huidige totale technische potentieel voor bio-energieproductie in de semi-aride en aride gebieden van de acht landen is berekend op ongeveer 300 PJ y^{-1} voor cassave-ethanol, 600 PJ y^{-1} voor jatropha-biodiesel, en 4.000 PJ y^{-1} voor brandhout. Verder bleek dat in veel semi-aride gebieden, cassave-ethanol, jatropha-olie, en brandhout economisch kan concurreren met de referentie-energiebronnen (benzine, diesel, en brandhout). De productiekosten van brandhout, jatropha-olie en cassave-ethanol zijn in de meeste aride gebieden echter hoger dan de gemiddelde nationale marktprijzen voor brandhout, diesel en benzine. Vanwege de mogelijke stimulans voor economische en sociale ontwikkeling van het platteland is het, ondanks de hoge productiekosten, belangrijk om duurzame bio-energieproductie in semi-aride en aride gebieden van Sub-Sahara Afrika te onderzoeken en hierin te investeren.

Hoofdstuk 5 evalueert de omvang en locatie van verzilte gronden wereldwijd, het huidige gebruik van deze gronden en het bestaande type landbedekking, alsmede de technische en economische potenties van biomassaproductie middels bosbouw op deze gronden (zogenaamde biosaline bosbouw). De wereldwijde omvang van verzilt land bedraagt ongeveer 1,1 Gha. 14% hiervan is geclassificeerd als bos, moerasland, of (inter-)nationaal beschermd gebied en wordt beschouwd als niet beschikbaar voor de productie van biomassa. Voor de resterende verzilte gronden (1,0 Gha) vond deze studie een gemiddelde biomassaopbrengst van 3,1 ton droge stof ha^{-1} y^{-1} en een wereldwijd technisch potentieel voor bio-energieproductie van 56 EJ y^{-1}. Dit is gelijk aan circa 11% van de huidige mondiale primaire energieconsumptie. Als ook bestaand landbouwgrond als niet-beschikbaar wordt beschouwd omwille van de duurzaamheid, dan daalt het technische potentieel tot 44 EJ y^{-1}. Het wereldwijde economische potentieel van biosaline bosbouw voor productiekosten van 2 € GJ^{-1} of minder werd berekend op 21 EJ y^{-1} (inclusief landbouwgrond), respectievelijk 12 EJ y^{-1} (exclusief landbouwgrond). Bij productiekosten van maximaal 5 € GJ^{-1} stijgt het wereldwijde economische potentieel tot 53 EJ y^{-1} (inclusief landbouwgrond), respectievelijk 39 EJ y^{-1} (exclusief landbouwgrond). Biosaline bosbouw kan een belangrijke bijdrage leveren aan het voorzien in de energievraag in bepaalde regio's, zoals bijvoorbeeld Afrika. Biosaline bosbouw heeft ook vele andere voordelen die het interessant maken hier verder onderzoek naar te doen en er ook in te investeren, zoals de mogelijkheid de kwaliteit van verzilte grond te verbeteren, inkomsten te genereren uit gronden die eerder laagproductief of onproductief waren, en koolstof vast te leggen.

Hoofdstuk 6 analyseert de netto contante waarde (net present value, NPV), de energieproductiekosten, en de milieu-effecten (vooral het effect op de uitstoot van broeikasgassen) van verschillende biosaline land- / bosbouwsystemen in drie contexten in Zuid Azië. De handel in carbon credits die gegenereerd kunnen worden door biosaline land- en bosbouw, is als een mogelijke extra bron van inkomsten meegenomen. De NPV bij een discontovoet van 10% is 1,0 k€ ha^{-1} voor een land- / bosbouwsysteem met rijst en bomen op verzilte (saline) gronden in de kustgebieden van Bangladesh (case studie 1); 4,8 k€ ha^{-1} voor een land- / bosbouwsysteem met rijst, tarwe en bomen op verzilte (sodic topsoils / saline-sodic subsoils) bodems in de staat Haryana in India (case studie 2); en 2,8 k€ ha^{-1} voor een kleinschalige *Acacia nilotica* plantage op verzilte (saline-sodic) grond in de provincie Punjab in Pakistan (case studie 3). De broeikasgasbalans van de drie systemen toont koolstofvastlegging aan van 16 ton CO_2-eq. ha^{-1} in Bangladesh, 26 t CO_2-eq. ha^{-1} in India, en 96 ton CO_2-eq. ha^{-1} in Pakistan gedurende de levensduur van de plantages. Dit vertaalt zich in een verhoging van de NPV met 3-80% in case studie 1, 1-14% in case studie 2 en 9-129% in case studie 3, afhankelijk van de prijs van de carbon credits die in dit onderzoek is verondersteld (1-15 € t CO_2-eq.). Hoewel de NPV in de drie case studies positief is, gaf de analyse aan dat de economische prestaties sterk afhankelijk zijn van het type en de ernst van de verzilting (omdat die het type en samenstelling van het land- / bosbouwsysteem evenals de geschikte boomsoorten beïnvloeden), de omlooptijd die wordt gerealiseerd, de verschillende markten voor houtproducten, de mogelijkheid tot de handel in carbon credits, en de discontovoet. Een eenvoudige extrapolatie van de resultaten suggereert een technisch potentieel van bio-energieproductie van 1,2 EJ y^{-1} (ongeveer 3% van de huidige totale primaire energieconsumptie van de drie landen) en een mitigatiepotentieel voor broeikasgasemissies van 43 miljoen ton CO_2-eq. die jaarlijks kan worden vastgelegd (ongeveer 3% van de uitstoot van broeikasgassen gerelateerd aan energieverbruik in de drie landen).

Gebaseerd op de bevindingen van de hoofdstukken 2-6, kunnen de volgende antwoorden op de onderzoeksvragen worden gegeven en aanbevelingen voor toekomstig onderzoek worden gedaan:

I Wat is het potentieel van bio-energieproductie op gedegradeerde en marginale gronden binnen verschillende contexten en op verschillende geografische schaalniveaus?

Het technisch potentieel van bio-energieproductie op gedegradeerde en marginale gronden is in dit proefschrift binnen verschillende contexten en op verschillende geografische schaalniveaus geanalyseerd. Tabel 7.2 geeft een overzicht van de gevonden resultaten in de hoofdstukken 3 tot en met 6. De uitkomsten wijzen op een mondiaal technisch potentieel van ongeveer 56 EJ y^{-1} op verzilt land (tabel 7.2). Combinatie van dit potentieel met het potentieel van 32 EJ y^{-1} bio-energie op door de mens veroorzaakte gedegradeerde gronden (met uitzondering van verzilt land) zoals geschat door Nijsen *et al.* (zie ook paragraaf 1.2.1), geeft aan dat het technisch potentieel van bio-energieproductie op gedegradeerde en marginale gronden ongeveer 90 EJ y^{-1} bedraagt. Dit staat gelijk aan ongeveer 18% van het huidige wereldwijde gebruik van primaire energie van circa 514 EJ y^{-1} (OECD/IEA, 2010c). Een vergelijking met resultaten die gevonden zijn in de literatuur, geeft aan dat dit potentieel ligt tussen de schatting van Dornburg *et al.* (2010, 70 EJ y^{-1}) en de bovenkant van de bandbreedte zoals gegeven door Hoogwijk *et al.* (2003, 8-110 EJ y^{-1}) en Schubert *et al.* (2009, 34-120 EJ y^{-1}). Het meenemen van mogelijke bio-energieproductie op niet-verzilte aride en semi-aride gebieden in Sub-Sahara Afrika en elders, kan leiden tot een hogere schatting van dit potentieel.

Er zijn echter ook factoren die slechts in beperkte mate in het onderzoek naar het bio-energie productiepotentieel van gedegradeerde en marginale gronden zijn meegenomen. De twee belangrijkste factoren zijn de mogelijke verergering van watertekorten in waterarme regio's en het huidige gebruik en de huidige functie van gedegradeerde en marginale gronden. Beide factoren kunnen het genoemde potentieel verminderen, maar de invloed van deze factoren op het daadwerkelijke duurzame potentieel is op dit moment onduidelijk.

Met betrekking tot water is meer onderzoek nodig naar de hydrologische effecten van (grootschalige) bio-energieproductie op verschillende geografische schaalniveaus; de economische, sociale en milieu-effecten van een toegenomen concurrentie om water; de potentieel positieve effecten van meerjarige bio-energiegewassen op waterinfiltratie en waterretentie van de bodem; en hoe deze aspecten het duurzame potentieel van bio-energieproductie op gedegradeerde en marginale gronden beïnvloeden. Voor een dergelijke beoordeling moeten ruimtelijk-expliciete datasets over het grondwaterpeil en de grondwaterkwaliteit beschikbaar zijn. Echter, zulke datasets met gegevens op een mondiale en continentale schaal bestaan niet, zoals is gebleken bij het onderzoek naar bio-energieproductie op verzilt land (zie hoofdstuk 5). Verder onderzoek moet zich dus eerst richten op het genereren van (meer) gegevens over het grondwater op nationaal niveau, zodat ook gedetailleerde datasets op continentaal en mondiaal niveau kunnen worden gemaakt.

Tabel 7.2: Overzicht van het technische potentieel van bio-energieproductie op gedegradeerde en marginale gronden binnen verschillende contexten en op verschillende geografische schaalniveaus, waarbij rekening is gehouden met beperkingen vanwege de noodzaak van duurzaam landgebruik.

Hoofdstuk	Context	**Geografisch schaalniveau**	**Omvang**		**Gemiddelde opbrengst**		**Potentieel**	
			Mha		$t\ ha^{-1}\ y^{-1}$ [a]		$EJ\ y^{-1}$	
3	- Palmolieproductie op *Imperata* graslanden	Maleisië	1		4,3	6,1[b]	0,1	0,2[b]
		Indonesië	12		3,5	5,9[b]	1,1	1,9[b]
4	- Cassave-ethanol, jatropha-olie, en brandhoutproductie op marginale semi-aride en aride gronden	Nationaal: Botswana	8	4[c]	5,5	0,7[c]	0,8	0,1[c]
		Burkina Faso	2	0	10,0	6,3	0,3	1,0
		Kenia	3	3	12,4	8,9	0,7	0,5
		Mali	3	6	8,1	2,7	0,5	0,5
		Senegal	1	1	7,4	5,7	0,1	0,1
		Zuid-Afrika	1	0	8,7	1,1	0,1	0
		Tanzania	2	0	12,4	n.v.t.	0,5	0
		Zambia	2	0	9,5	n.v.t.	0,3	0
		- Subcontinentaal: acht landen in Sub-Sahara Afrika samen	22	13	7,2	4,1	3,2	1,1
5	- Houtige biomassaproductie van bosbouw op verzilte bodems	Mondiaal	971		3,1		56,2[d]	
		Regio's met de grootste potenties:						
		Oceanië	144		7,6		20,2	
		Voormalige USSR	117		4,7		10,0	
		Zuid-Amerika	57		5,2		5,4	
6	- Houtige biomassaproductie van land- / bosbouw op verzilte bodems	Regionaal: Bangladesh, India en Pakistan samen	11		6,0		1,2	

a - Gemiddelde opbrengst gegeven voor palmolie in ton ruwe palmolie (crude palm oil, CPO) $ha^{-1}\ y^{-1}$ en voor brandhout en houtige biomassa in ton droge stof (t dm) $ha^{-1}\ y^{-1}$.

b - De opbrengsten in 2020 zijn vermeld voor twee scenario's: *basis scenario* (linker kolom) en *verbeterd scenario* (rechtse kolom).

c - De omvang, de gemiddelde opbrengst, en de potentiëlen worden alleen getoond voor semi-aride (linker kolom) en aride (rechter kolom) regio's voor brandhout. Opbrengsten en potentiëlen van cassave-ethanol en jatropha-olie zijn vermeld in tabel 4.1 en tabel 4.4 (hoofdstuk 4).

d - De omvang, de gemiddelde opbrengst, en het energiepotentieel van houtige biomassaproductie op verzilt land omvatten ook deels landbouwgrond. De resultaten voor het energiepotentieel exclusief het gebruik van bestaand landbouwgrond, zijn weergegeven in hoofdstuk 5.

n.v.t. - Niet van toepassing

Vanwege het huidige gebruik en de functie van gedegradeerde en marginale gronden, heeft dit proefschrift een poging gedaan om gebieden met belangrijke concurrerende toepassingen en functies – te weten de productie van levensmiddelen en diervoeding en het behoud van een hoge of unieke biodiversiteit - uit te sluiten. Echter, de toegepaste datasets houden niet met alle toepassingen rekening. Voorbeelden van landgebruik en functies die bijzonder moeilijk zijn om nauwkeurig vast te leggen, zijn het gebruik voor veeteelt, voor jagers en verzamelaars, voor het behoud ecosysteem functies en voor culturele waarden. Het niet goed verwerken van deze toepassingen en functies kan resulteren in verplaatsing van bio-energieproductie naar een ander gebied en in te hoge schattingen voor het potentieel van bio-energieproductie. Daarom is het nodig dat toekomstig onderzoek deze toepassingen en diensten analyseert en de invloed daarvan op de beschikbaarheid van gedegradeerd en marginaal land voor bio-energieproductie beter in kaart brengt. Bovendien moet bij de planning of het beoordelen van concrete projecten een evaluatie worden gemaakt van het huidige landgebruik, het eigendom, en functies die thans worden vervuld, om niet-duurzaam landgebruik, verlies van ecosysteemfuncties, en negatieve sociale en ecologische gevolgen van bio-energieproductie tegen te gaan.

II Hoe ziet de economie van bio-energieproductie op deze gronden eruit en wat zijn mogelijk bijkomende voordelen?

De resultaten van dit proefschrift laten zien dat biomassa en bio-energieproductie op gedegradeerde en marginale gronden economisch haalbaar kan zijn en kan bijdragen aan de lokale en regionale behoefte aan biomassa en/of energie. Hoofdstuk 6 analyseert de economische prestaties van land- / bosbouwsystemen op verschillende soorten verzilte gronden in Zuid Azië, in termen van netto contante waarde (NPV). Hoewel de NPV positief is in de drie onderzochte case studies, blijkt uit de analyses ook dat de economische prestaties sterk afhangen van het type en de ernst van de verzilting (omdat die het type en de samenstelling van het land- / bosbouwsysteem en de te planten boomsoorten beïnvloeden), de omlooptijd, de markten voor houtproducten, de mogelijkheid tot de deelname aan emissiehandel, en de discontovoet. Door het meenemen van de economische waarde van koolstofvastlegging in de (land-) bosbouwsystemen kan de NPV stijgen met maximaal zo’n 129% t.o.v. de situatie zonder CO_2 emissiehandel, afhankelijk van de CO_2-prijs die verondersteld wordt in de analyse (in dit proefschrift: 1-15 € t^{-1} CO_2-eq.) en van de vraag of het land- / bosbouwsysteem in aanmerking komt voor emissiehandel. Verder vergen de toelatingseisen en praktische hindernissen voor kleine boeren die willen deelnemen aan emissiehandel schema's aandacht.

De kosten van de verschillende bio-energie productiesystemen blijken te kunnen concurreren met relevante marktprijzen. Voorbeelden van deze systemen zijn brandhoutproductie in veel semi-aride gebieden in Sub-Sahara Afrika, jatropha-olie en cassave-ethanol productie in een beperkt aantal semi-aride gebieden in Sub-Sahara Afrika (hoofdstuk 4) en brandhout- en houtskoolproductie in land- / bosbouw plantages op verzilte gronden in Zuid Azië (hoofdstuk 6). Echter, in de meeste aride gebieden in Sub-Sahara Afrika zijn de productiekosten van brandhout, jatropha-olie, en cassave-ethanol vaak hoger dan de gemiddelde nationale marktprijzen van benzine, diesel en brandhout. Ondanks de hoge productiekosten, kan duurzame bio-energieproductie in deze regio nog steeds wenselijk zijn, omdat het een bijdrage kan leveren aan rurale economische en sociale ontwikkeling. Mogelijke voordelen van deze rurale ontwikkeling bestaan uit het verhogen van meer lokaal geproduceerde energiebronnen op het platteland, het creëren van nieuwe markten voor landbouwproducten, en het generen van meer inkomen voor de plattelandsbevolking.

Gebaseerd op de berekende productiekosten, werden aanbodcurven gemaakt om het economisch potentieel van bio-energieproductie op gedegradeerde en marginale gronden te bepalen, zie Figuur 4.4 en Figuur 5.4 van dit proefschrift. De resultaten tonen aan dat het economisch potentieel significant is voor vele regio's. Bijvoorbeeld, hoofdstuk 4 laat een economische potentieel van brandhoutproductie in semi-aride gebieden in Tanzania zien van 103 PJ y^{-1} bij productiekosten van maximaal 2 € GJ^{-1}. Dit is gelijk aan 16% van het huidige primaire energiegebruik van Tanzania. Een ander voorbeeld is te vinden in hoofdstuk 5, waar werd aangetoond dat bio-energieproductie op verzilte gronden (biosaline bosbouw) kan bijdragen aan de energievoorziening, in het bijzonder aan die van Afrika; het economisch potentieel is berekend op 8 EJ y^{-1} bij productiekosten van maximaal 2 € GJ^{-1}. Dat is ongeveer 28% van de huidige totale primaire energieconsumptie in Afrika. Het wereldwijde economische potentieel van biomassaproductie op verzilte gronden, met inbegrip van landbouwgrond, is bepaald op 21 EJ y^{-1} of 4% van het mondiale primaire energiegebruik) bij productiekosten van 2 € GJ^{-1} of minder. Het economisch potentieel neemt toe tot 53 EJ y^{-1} als de productiekosten voor biomassa mogen toenemen tot 5 € GJ^{-1}. Wanneer, vanwege duurzaamheidsredenen, het gebruik van bestaand landbouwgrond wordt uitgesloten, daalt het economisch potentieel van biosaline bosbouw tot 12 EJ y^{-1} bij productiekosten van 2 € GJ^{-1} of minder, en tot 39 EJ y^{-1} bij productiekosten van 5 € GJ^{-1} of minder.

III Wat zijn milieu-effecten van bio-energieproductie op gedegradeerde en marginaal gronden en hoe hangen deze af van de context waarbinnen de energieproductie gebeurt?

Uit dit proefschrift blijkt dat bio-energieproductie uit houtige gewassen van gedegradeerde en marginale gronden zowel positieve als negatieve milieu-effecten kan hebben. Mogelijke positieve effecten zijn koolstofvastlegging, een toename van de bodemvruchtbaarheid, verbeterde waterinfiltratie en waterretentie van de bodem, vermindering tot omkering van bodemdegradatie, en verlaging van het zoutgehalte van de bodem. Gedetailleerde analyses van de broeikasgasbalans voor twee voorbeelden van bio-energieproductie op gedegradeerde en marginale gronden – de op palmolie gebaseerde energieproductie op *Imperata* grasland in Maleisië (hoofdstuk 2) en de productie van biomassa via land- / bosbouwsystemen op verzilte gronden in Zuid Azië (hoofdstuk 6) – laten zien dat deze bio-energieproductie een emissiereductie van meer dan 100% kan bereiken ten opzichte van het gebruik van fossiele brandstoffen en dat bio-energieplantages op gedegradeerd en marginaal land kunnen veranderen in koolstofvastlegging systemen. Uit hoofdstuk 2 blijkt verder dat verandering van landgebruik de doorslaggevende factor is in de berekening van de totale uitstoot van broeikasgassen van palmolie-energieproductie. Verder blijkt dat indien natuurlijke regenwouden of veengronden worden gebruikt in plaats van gedegradeerd land, de emissiereductiedoelstellingen van de Renewable Energy Directive van de Europese Commissie niet kunnen worden gehaald. Echter, als de productie van palmolie plaatsvindt op gedegradeerde gronden en het plantage-beheer verbeterd, kan wel worden voldaan aan de gestelde eis voor emissiereductie, en kan elektriciteit op basis van palmolie vanuit het oogpunt van de broeikasgasbalans als duurzaam worden beschouwd.

Ondanks deze mogelijke positieve effecten, kan bio-energieproductie op gedegradeerde en marginale gronden ook risico's voor bodem, biodiversiteit en water met zich meebrengen. De analyse van de land- /bosbouwsystemen op verzilte gronden in Zuid Azië (hoofdstuk 6) geeft bijvoorbeeld aan dat er een risico bestaat dat boomsoorten die tolerant zijn voor de moeilijkere groeiomstandigheden op gedegradeerde en marginale gronden, zich invasief kunnen gedragen en zich kunnen verspreiden, zelfs op niet gedegradeerde of marginale gronden. Om dit risico te minimaliseren, wordt aanbevolen dat voornamelijk inheemse soorten worden geplant en soort-specifiek beheer wordt toegepast. Een ander risico is de mogelijke verergering van watertekorten in de regio's waar water al schaars is. Zoals aangegeven in het antwoord op onderzoeksvraag 1, is meer onderzoek nodig naar de economische, sociale en ecologische (met name hydrologische) effecten van een toenemende vraag naar water door (grootschalige) bio-energieproductie op gedegradeerde en marginale gronden. Daarnaast is er meer systematisch onderzoek nodig om de omstandigheden te bepalen (bv. type en de ernst van de degradatie) waaronder bio-energieproductie lokaal negatieve effecten kan hebben op de biodiversiteit, zodat deze kunnen worden vermeden. Bovendien benadrukt hoofdstuk 4 dat het verzamelen van gedetailleerde biodiversiteits-data op nationaal niveau belangrijk is om beter rekening te kunnen houden met biodiversiteit in de becijfering van het potentieel van bio-energie.

Ook sociale gevolgen en de socio-economische effecten van bio-energieproductie op gedegradeerde en marginale gronden moeten nader worden onderzocht. Twee belangrijke aspecten hierbij zijn: 1) de mogelijk (in)directe veranderingen in landgebruik veroorzaakt door het gebruik van gedegradeerd en marginaal land voor bio-energieproductie en de bijbehorende kansen (met betrekking tot bijvoorbeeld het genereren van inkomsten uit laagproductieve grond en de regeneratie van bodems) en risico's (in verband met bijvoorbeeld de verplaatsing van jagers/verzamelaars en nomadische veeteelt-gemeenschappen), en 2) de voorwaarden waaronder deze risico's kunnen worden geminimaliseerd.

Samenvattend kunnen drie belangrijke conclusies worden getrokken. Ten eerste, bio-energieproductie op gedegradeerde en marginale gronden heeft een aanzienlijk potentieel dat kan bijdragen aan het mondiale energieverbruik (ongeveer 18% van de huidige mondiale verbruik van primaire energie). Ten tweede, de economische analyses suggereren dat de productie van bio-energie op deze gronden in veel gevallen kan concurreren met andere bio-energiebronnen, alsmede met fossiele brandstoffen. Ten derde, productie van meerjarige bio-energiegewassen op gedegradeerde en marginale gronden kan zowel positieve (bijvoorbeeld koolstofvastlegging, verbeteringen in de vruchtbaarheid van de bodem en een verlaging van bodemdegradatie) en negatieve (bijvoorbeeld verergering van watertekorten in waterarme gebieden) milieu-effecten hebben. Meer onderzoek is nodig om de omstandigheden te bepalen waaronder de productie van bio-energie resulteert in negatieve milieu-effecten, zodat deze kunnen worden vermeden. Gezien deze bevindingen, zijn de volgende aanbevelingen voor verder onderzoek geïdentificeerd:

1) In verschillende contexten moeten demonstratieprojecten worden uitgevoerd om ervaring op te doen met het opzetten en beheren van duurzame bio-energieproductie op gedegradeerde en marginale gronden.
2) De ontwikkeling van infrastructuur, en capaciteitsopbouw is nodig. De opbouw van capaciteit is vooral noodzakelijk met betrekking tot het beheer van duurzame bio-energieproductie in verschillende situaties van gedegradeerde en marginale gronden en voor het minimaliseren van mogelijke negatieve milieu-effecten.
3) De potentiële, bijkomende voordelen van de productie van meerjarige bio-energiegewassen - zoals de mogelijkheid om gedegradeerd land te herstellen, het genereren van inkomsten uit eerder laagproductieve of onproductieve grond, koolstofvastlegging, het verbeteren van de waterretentie van de bodem, en erosiecontrole - zijn belangrijke redenen voor verder onderzoek naar, en investering in bio-energieproductie op gedegradeerde en marginale gronden. Het waarderen van de economische waarde van deze bijkomende voordelen in economische evaluaties kan bio-energieproductie op deze gronden aantrekkelijker maken. Stimulering en beleidsmaatregelen zijn nodig voor het internaliseren van deze externe effecten.
4) Hoewel de productie van bio-energie op gedegradeerde en marginale gronden positieve milieu-effecten kan hebben, moet ook worden voldaan aan duurzaamheidscriteria op andere gebieden. Dit vereist volwaardige certificering, net als bij de productie van bio-energiedragers op ander typen land.

Kapitel 7

Zusammenfassung und Schlussfolgerungen

Das heutige globale Energiesystem ist in erster Linie gekennzeichnet von fossilen Brennstoffen, deren Verwendung weithin als nicht nachhaltig gilt im Zusammenhang mit den hohen Treibhausgasemissionen (THG-Emissionen) in die Atmosphäre, der Endlichkeit der fossilen Brennstoffe, der ungleichen geografischen Verteilung von fossilen Brennstoffen, und dem Versagen des Systems, Milliarden von Menschen mit Zugang zu modernen Energiedienstleistungen zu versorgen. Bioenergie dagegen ist eine wichtige Option, um ein zukünftiges Energiesystem nachhaltiger zu gestalten. Zum einen hat Bioenergie ein großes Wachstumspotenzial und könnte daher einen wichtigen Beitrag zur zukünftigen Energieversorgung leisten. Zweitens kann Bioenergie, wenn nachhaltig produziert, im Vergleich zu fossilen Brennstoffen zur Senkung der Treibhausgasemissionen führen. Drittens ist die Bioenergie ein vielseitiger Energieträger, der für die Erzeugung von Wärme und Elektrizität sowie feste, flüssige und gasförmige Brennstoffe verwendet werden kann. Viertens sind Bioenergieressourcen weltweit gleichmäßiger verteilt als fossile Brennstoffe; dies verringert die Abhängigkeit von Energieimporten aus einer kleinen Anzahl von Ländern und erhöht die lokale Energieproduktion.

Allerdings ist der zunehmende globale Handel und Konsum von Bioenergie in Industrieländern durch eine wachsende Besorgnis über die umweltbezogenen, ökologischen und sozialen Auswirkungen der (modernen) Bioenergieproduktion begleitet worden. Viele dieser unbeabsichtigten und unerwünschten Auswirkungen der Bioenergieerzeugung stehen in Zusammenhang mit direkten Landnutzungsänderungen (land use change, LUC; also die geänderte Flächennutzung) und indirekten LUC (also die durch die Ausdehnung der Bioenergieerzeugung geänderte Flächennutzung an einem Ort induziert Landnutzungsänderungen an einem anderen Ort). Die Erzeugung von Bioenergie auf degradierten oder marginalen Flächen bietet demgegenüber Vorteile, weil diese Flächen weitgehend ungeeignet und oft wirtschaftlich unattraktiv für landwirtschaftliche Produktion sind. Die Produktion von mehrjährigen Energiepflanzen auf degradierten und marginalen Flächen kann auch Kohlenstoff speichern, die Bodenfruchtbarkeit verbessern und andere Bodendegradationsprozesse, wie Erosion, Dispersion und Auslaugung, als Folge des ober- und unterirdischen Biomassewachstums reduzieren. Darüber hinaus können mehrjährige Energiepflanzen auf degradierten und marginalen Flächen die Artenvielfalt erhöhen, besonders wenn dadurch Monokulturen und große Felder vermieden werden und eine Mischung aus verschiedenen bodendeckenden Pflanzen kultiviert wird. Darüber hinaus kann die Erzeugung von Bioenergie auf degradierten und marginalen Flächen zur sozialen und wirtschaftlichen Entwicklung des ländlichen Raumes beitragen, indem Land mit geringer oder ohne vorherige Produktivität nutzbar gemacht wird.

Trotz seiner möglichen Vorteile hat der Anbau von Energiepflanzen auf degradierten und marginalen Flächen auch Nachteile; diese können die Wirtschaftlichkeit der Bioenergieproduktion einschränken und ihre Nachhaltigkeit verringern. Die wichtigsten Herausforderungen beziehen sich auf 1) die schwierigen Anbaubedingungen, die einen großen Aufwand über einen langen Zeitraum erfordern und oft zu einem geringeren Ertrag als auf Flächen mit hoher Produktivität führen und 2) dass degradierte und marginale Flächen oft eine wichtige Ressource für arme, ländliche Gemeinden sind, insbesondere derjenigen, die nicht-gesichertete Landrechte besitzen. Darüber hinaus können degradierte und marginale Flächen noch gewisse Ökosystemfunktionen und Biodiversitätsniveaus unterstützen, die denen von landwirtschaftlich genutzten Flächen ähnlich sind; Erhaltung oder Erhöhung dieser Funktionen und Werte stellt eine weitere, mögliche Herausforderung dar. Während diese Herausforderungen in vielen Studien anerkannt werden, ist wenig über die Auswirkungen auf die technischen und wirtschaftlichen Potenziale, die Wirtschaftlichkeit und die ökologischen Auswirkungen der Bioenergieproduktion auf degradierten Flächen bekannt. Vor allem fehlt es an Wissen über die Vorraussetzungen, unter denen ökologische und soziale Risiken der Nutzung degradierter und marginaler Flächen für die Bioenergieproduktion bestehen und wie diese Risiken vermieden werden können. Darüber hinaus ist mehr Forschung notwendig, um das tatsächliche Potenzial degradierter und marginaler Flächen zur Deckung des globalen, regionalen, nationalen und lokalen (Bio-)Energiebedarfs bestimmen zu können. Die Berechnung der Potenziale von Bioenergieproduktion auf degradierten und marginalen Flächen auf verschiedenen geografischen Ebenen sollte sich daher auf das Ausmaß und den Schweregrad von degradierten und marginalen Flächen, ihre Verfügbarkeit und Eignung für die Bioenergieproduktion und die Erträge konzentrieren. Darüber hinaus sind die Kosten von Biomasseproduktion auf degradierten und marginalen Flächen wichtig um festzustellen, ob diese Bioenergieproduktion wirtschaftlich ist. Weiterhin ist detaillierteres Wissen in Bezug auf die Produktionskosten und das wirtschaftliche Potenzial der Bioenergieproduktion auf verschiedenen Arten und Schweregraden von degradierten und marginalen Flächen erforderlich.

Ziel dieser Dissertation war es, einige dieser Wissenslücken zu schließen. Das Hauptziel bestand in der Beurteilung der Potenziale, der Wirtschaftlichkeit und der ökologischen Auswirkungen der Bioenergieproduktion auf degradierten und marginalen Flächen für unterschiedliche Rahmenbedingungen und geografische Ebenen (von lokal bis global). Zu diesem Zweck wurden folgende Fragestellungen bearbeitet:

I Wie hoch sind die technischen Potenziale für die Erzeugung von Bioenergie auf degradierten und marginalen Flächen, wenn unterschiedliche Rahmenbedingungen und verschiedene geografische Ebenen betrachtet werden?

II Ist die Erzeugung von Bioenergie auf degradierten und marginalen Flächen unter verschiedenen Rahmenbedingungen, und gegenenfalls unter Berücksichtigung positiver Nebeneffekte, wirtschaftlich?

III Welche ökologischen Auswirkungen hat die Bioenergieproduktion, unter Annahme verschiedener Rahmenbedingungen, auf degradierten und marginalen Flächen?

Diese Fragestellungen wurden in Kapitel 2 bis 6 behandelt. Alle Kapitel analysieren Bioenergieproduktion auf degradierten und marginalen Flächen, allerdings für unterschiedliche Rahmenbedingungen und geographischen Ebenen. Kapitel 2 und 3 konzentrieren sich auf die Palmölproduktion in Malaysia und Indonesien aufgrund der weltweiten Debatte über die negativen Umweltauswirkungen von Palmöl. Diese Kapitel bewerten die Nutzung von *Imperata* Grasland als Alternative zur Abholzung von tropischem Regenwald oder zu anderen Landtypen. Kapitel 4 bewertet die Produktion von Ethanol aus Maniok, Jatropha-Öl und Brennholz auf marginalen semi-ariden und ariden Flächen in acht afrikanischen Ländern südlich der Sahara, da semi-aride und aride Gebiete in dieser Hinsicht bisher zu wenig berücksichtigt wurden und andere Bedingungen für und Anforderungen an die Bioenergieerzeugung haben als Gebiete mit mehr Niederschlag. Kapitel 5 und 6 untersuchen die Erzeugung von holzartiger Biomasse auf versalzenen Flächen mit Schwerpunkt auf der globalen Ebene (Kapitel 5) und auf den lokalen und nationalen Ebenen in Bangladesch, Indien und Pakistan (Kapitel 6). Versalzene Flächen sollten wegen ihrer großen Ausdehnung und den Schwierigkeiten, die sie für die Landwirtschaft darstellen, berücksichtigt werden.

Tabelle 7.1 gibt einen Überblick über die Rahmenbedingungen, geografischen Ebenen, und Fragestellungen, die in dieser Dissertation behandelt wurden. Daraufhin folgen eine Zusammenfassung der einzelnen Kapitel und ihrer wichtigsten Ergebnisse und die Antworten auf die drei übergreifenden Fragestellungen.

Tabelle 7.1: Übersicht über die Rahmenbedingungen der Bioenergieproduktion auf degradierten und marginalen Flächen, die geographischen Ebenen und die Fragestellungen, die in den Kapiteln 2 bis 6 behandelt werden

Kapitel	**Rahmenbedingungen**	**Geographische Ebenen**	**Fragestellung**		
			I	II	III
2	- Palmölproduktion auf *Imperata* Grasland und anderen Flächen	- Lokal (Fallstudie in Nord-Borneo, Malaysia)			•
3	- Landnutzungsänderung und Palmölproduktion auf *Imperata* Grasland	- National (Indonesien und Malaysia)	•		•
4	- Produktion von Maniok-Ethanol, Jatropha-Öl und Brennholz auf marginalen semi-ariden und ariden Flächen	- Subkontinental - national (acht Länder in Subsahara-Afrika)	•	•	
5	- Erzeugung von holzartiger Biomasse auf forstwirtschaftlichen Plantagen auf versalzenen Flächen	- Global - subkontinental (17 Weltregionen)	•	•	
6	- Erzeugung von holzartiger Biomasse und Bioenergie auf (agro-)forstwirtschaftlichen Plantagen auf versalzenen Flächen	- Subkontinental - lokal (drei Fallstudien in Südasien)		•	•

Kapitel 2 analysiert die THG-Emissionen, die bei der Produktion von rohem Palmöl (crude palm oil, CPO) und Palmfettsäure Destillat (palm fatty acid distillate, PFAD) in Nord-Borneo (Malaysia), dem Transport dieser Erzeugnisse in die Niederlande und ihrer Mitverbrennung mit Erdgas für die Stromerzeugung freigesetzt werden. Im Falle des CPO wurde ebenfalls die Umsetzung in Biodiesel und die damit verbundenen Treibhausgasemissionen untersucht. Die Ergebnisse zeigten, dass Landnutzungs-änderungen der entscheidende Faktor in der Treibhausgasbilanz waren und dass Palmöl-Energieketten basierend auf ehemaligen tropischen Regenwäldern oder Feuchtgebieten solch große Emissionen haben, dass sie das durch die Europäische Kommission in der Richtlinie Erneuerbare Energien festgelegte Reduktionsziel nicht erfüllen können. Wenn die CPO Produktion allerdings auf degradierten Flächen erfolgt und das Plantagenmanagement verbessert wird, oder wenn das Nebenprodukt PFAD für die Stromerzeugung verwendet wird, können die Emissionsminderungskriterien erfüllt und Palmöl-Strom in puncto THG-Emissionen als nachhaltig betrachtet werden. In unserem Basisszenario (ehemaliger Naturwald, in dem Nutzholz selektiv entfernt wurde, logged-over forest) kann Biodiesel aus Palmöl die THG-Emissionen um 30% reduzieren. Unter anderen Rahmenbedingungen, wie z.B. Palmölproduktion auf degradierten Flächen und ein besseres Plantagenmanagement kann jedoch eine Emissionsminderung von mehr als 150% erreicht werden. Dies würde Palmölplantagen zu Kohlenstoffspeichern machen. Dieses Kapitel schlussfolgert daher, dass degradierte Flächen für die Palmölproduktion verwendet und das

Plantagenmanagement verbessert werden sollten, um Bio-Strom und Biodiesel nachhaltig aus Palmöl und dessen Derivaten herstellen zu können.

Aufgrund des starken Einflusses der LUC auf die Treibhausgasbilanz von Palmöl-Energieketten, stellt **Kapitel 3** Daten über LUC auf nationaler Ebene in Indonesien und Malaysia zusammen und analysiert Ausmaße und Ursachen der vergangenen 30 Jahre. Dieses Kapitel untersucht dabei auch den Anteil, den Palmöl in der Vergangenheit an LUC hatte und welche Rolle das vorraussichtliche Wachstum der Palmölproduktion bis 2020 in Bezug auf zukünftige LUC spielen kann. Weiterhin werden in diesem Kapitel Strategien empfohlen, um negative Auswirkungen zu minimieren. Die Datenerhebung für diese Studie ergab, dass die Qualität und Quantität der LUC-Daten auf nationaler Ebene in der Vergangenheit gering waren. Trotz dieser Unsicherheiten zeigte die Übersicht der bisherigen LUC, dass große Veränderungen der Landnutzung in Indonesien und Malaysia aufgetreten sind. In Indonesien ist LUC in erster Linie von der Abholzung des tropischen Regenwaldes von 40 Millionen Hektar (Mha) geprägt, was einem Verlust von 30% der Waldfläche entspricht. Die Entwaldung in Malaysia fiel sowohl absolut als auch relativ gesehen geringer aus: Der Verlust an tropischen Regenwaldflächen betrug fast 5 Mha, das entspricht einem Verlust von 20% der Waldfläche. Weitere große Veränderungen in Malaysia traten bei permanentem Ackerland (ohne Palmöl) auf, dessen Ausdehnung seit den frühen 1990er Jahren rasch zurückgegangen ist, sowie bei Palmölproduktionsflächen, die einen starken Anstieg erfahren haben. Projektionen für die Nachfrage nach zusätzlichen Flächen für die Palmölproduktion bis zum Jahr 2020 reichten von 1 bis 28 Mha in Indonesien. Diese Nachfrage kann zu einem großen Teil durch Anbau auf degradierten Flächen erfüllt werden, wenn keine weiteren Entwaldung angenommen wird. In Malaysia reichten die Expansionsprojektionen von 0,06 bis 5 Mha. Allerdings kann nur die Projektion mit dem niedrigsten Zuwachs realisiert werden, wenn nur degradierte Flächen verwendet werden dürfen. Die Rolle der Palmölproduktion in zukünftigen LUC hängt von der Größe der projizierten Expansion sowie von der landwirtschaftlichen Bewirtschaftung ab, wie z.B. der früheren Neubepflanzung mit ertragreicheren Pflanzen und der Etablierung neuer Plantagen auf degradierten Flächen. Allerdings muss die derzeitige Nutzung degradierter Flächen noch besser untersucht werden, um mögliche indirekte LUC, Grundbesitzkonflikte und andere soziale Auswirkungen zu verringern. Zusätzlich zur Minimierung der direkten und indirekten LUC durch den Palmölsektor, müssen auch Maßnahmen getroffen werden, die die Entwaldung reduzieren, die durch andere Ursachen ausgelöst wurde. Ein wesentliches Element darin ist eine bessere Planung und Steuerung der Flächennutzung. Dies beinhaltet eine angemessenere Abgrenzung von Wald und einen besseren Schutz von noch bewaldeten Flächen, ein verbessertes Monitoring der Landnutzung, und mehr Forschung, um die Komplexität und Dynamik der Ursachen und treibenden Kräfte von LUC aufzudecken.

Kapitel 4 analysiert die heutigen technischen und wirtschaftlichen Potenziale von drei Bioenergieproduktionssystemen (Maniok-Ethanol, Jatropha-Öl und Brennholz) in semi-ariden und ariden Regionen in acht afrikanischen Ländern südlich der Sahara: Botswana, Burkina Faso, Kenia, Mali, Senegal, Südafrika, Tansania und Sambia. Die Ergebnisse zeigten, dass die Verfügbarkeit von Flächen für die Bioenergieproduktion im Bereich von 2% (1,3 Mha) des gesamten semi-ariden und ariden Gebietes in Südafrika bis 21% (12 Mha) in Botswana lag. Die Verfügbarkeit von Flächen für die Bioenergieproduktion wird vor allem durch landwirtschaftliche Nutzung eingeschränkt, aber auch durch steiles Gelände und Schutz der biologischen Vielfalt. Das derzeitige technische Gesamtpotenzial für die semi-ariden und ariden Gebiete der acht untersuchten Länder wurde auf rund 300 PJ y^{-1} für Maniok-Ethanol, 600 PJ y^{-1} für Jatropha-Öl und 4.000 PJ y^{-1} für Brennholz berechnet. Die Analyse der wirtschaftlichen Potenziale zeigte, dass Maniok-Ethanol, Jatropha-Öl und Brennholz in vielen semi-ariden Regionen wirtschaftlich sein können im Vergleich zu den entsprechenden Referenzenergieträgern (Benzin, Diesel, und Brennholz). Allerdings sind die Produktionskosten aller drei Produkte in den meisten ariden Regionen Subsahra-Afrikas oft über den durchschnittlichen nationalen Marktpreisen für die Referenzenergieträger. Trotz der hohen Produktionskosten in manchen Regionen ist es wichtig, die nachhaltige Erzeugung von Bioenergie in semi-ariden und ariden Gebieten Subsahara-Afrikas zu untersuchen und in sie zu investieren, weil sie das Potenzial hat, die wirtschaftliche und soziale Entwicklung in ländlichen Gebieten voranzutreiben.

Kapitel 5 analysiert den Umfang und die geographische Lage von versalzenen Böden weltweit sowie deren heutige Landnutzung. Weiterhin werden die heutigen technischen und wirtschaftlichen Potenziale der Erzeugung von Biomasse aus forstwirtschaftlichen Plantagen auf diesen Böden (biosaline Forstwirtschaft) berechnet. Weltweit gibt es ca. 1,1 Milliarden Hektar (Gha) versalzene Flächen; davon sind 14% als Wald-, Feucht- oder (inter)nationale Schutzgebiete oder wegen Nachhaltigkeitsanforderungen als nicht für die Biomassekultivierung verfügbar eingestuft. Für die restlichen versalzenen Gebiete (1,0 Gha) fanden wir in diesem Kapitel einen durchschnittlichen Biomasseertrag von 3,1 t Trockenmasse ha^{-1} y^{-1} und ein globales technisches Potenzial von 56 EJ y^{-1} (entspricht 11% des heutigen globalen Primärenergieverbrauchs). Wenn landwirt-schaftliche Flächen wegen Nachhaltigkeitsanforderungen ebenfalls als nicht verfügbar betrachtet werden, dann verringert sich das technische Potenzial auf 44 EJ y^{-1}. Das globale wirtschaftliche Potenzial der biosalinen Forstwirtschaft bei Produktionskosten von 2 € GJ^{-1} oder weniger wurde auf ca. 21 EJ y^{-1} berechnet, wenn landwirtschaftlich genutzte Flächen berücksichtigt werden, und auf 12 EJ y^{-1} ohne diese Flächen. Bei Produktionskosten von bis zu 5 € GJ^{-1} erhöht sich das globale wirtschaftliche Potenzial auf 53 EJ y^{-1} (einschließlich landwirtschaftlich genutzter Flächen) und auf 39 EJ y^{-1} (ohne landwirtschaftlich genutzte Flächen). Biosaline Forstwirtschaft kann in bestimmten Regionen erheblich zur Energieversorgung beitragen, z. B. in Afrika. Außerdem hat biosaline Forstwirtschaft zahlreiche weitere Vorteile, die interessant für weitere Forschung und Investitionen sind: z.B. ihre potenzielle Verbesserung der Bodenfruchtbarkeit, die Erzeugung von Einkommen von bisher gering-produktiven oder unproduktiven Flächen sowie die Kohlenstoffspeicherung.

Kapitel 6 beurteilt die Wirtschaftlichkeit und die Umweltverträglichkeit unterschiedlicher biosaliner (agro-)forstwirtschaftlicher Systeme in drei Fallstudien in Südasien. Die

Wirtschaftlichkeit wird dabei anhand des Kapitalwerts (net present value, NPV) und der Produktionskosten beurteilt, die Umweltverträglichkeit vor allem über die Auswirkungen auf die Treibhausgasemissionen. Die wirtschaftlichen Effekte des Handels mit Emissionsgutschriften aus biosaliner (Agro-)Forstwirtschaft wurden dabei auch als mögliche zusätzliche Einnahmequelle bewertet. Der NPV bei einem Diskontsatz von 10% ist 1,0 k€ ha^{-1} für ein Agroforstwirtschaftssystem mit Reis und *Acacia nilotica* oder *Eucalyptus camladulensis* Bäumen auf versalzenen (saline) Böden in den Küstenregionen Bangladeschs (Fallstudie 1); 4,8 k€ ha^{-1} für ein Agroforstwirtschaftssystem mit Reis, Weizen und *Eucalyptus tereticornis* auf versalzenen (sodic / saline-sodic) Böden in Haryana, Indien (Fallstudie 2); und 2,8 k€ ha^{-1} für eine *Acacia nilotica* Baumplantage auf versalzenen (saline-sodic) Böden in der Provinz Punjab in Pakistan (Fallstudie 3). Die THG-Bilanz der drei Systeme zeigt eine Kohlenstoffspeicherung von 16 t CO_2-Äquivalente (CO_2 equivalent, CO_2-eq.) pro Hektar in Bangladesch, 26 t CO_2-eq. ha^{-1} in Indien, und 96 t CO_2-eq. ha^{-1} in Pakistan. Dies führt, je nach Wert der Emissionsgutschriften (1-15 € pro t CO_2-eq.), zu einer Erhöhung des NPV von 3-80% in Fallstudie 1, 1-14% in Fallstudie 2 und 9-129% in Fallstudie 3. Obwohl der NPV in den drei Fallstudien positiv ist, zeigte unsere Analyse, dass die Wirtschaftlichkeit stark von den folgenden Faktoren abhängt: 1) Art und Schweregrad der Versalzung: beides beeinflusst die Art des (Agro-)Forstwirtschaftssystems und die gepflanzten Baumarten sowie den Energieertrag, 2) Umtriebszeit, 3) Märkte und Entwicklung der Marktpreise für Produkte aus Holz, 4) die Möglichkeit mit Emissionszertifikaten zu handeln und 5) Diskontsatz. Eine einfache Extrapolation der Ergebnisse deutet auf ein technisches Potenzial von biosaliner (Agro-)Forstwirtschaft von 1,2 EJ y^{-1} Bioenergieproduktion hin (ca. 3% des gesamten heutigen Primärenergieverbrauchs der drei Länder, die in diesem Kapitel untersucht wurden) und eine potenzielle Emissionsminderung von 43 Mt CO_2-eq. (ca. 3% der Treibhausgasemissionen der drei Länder die aus deren Energieverbrauch entstehen).

Basierend auf den Erkenntnissen der Kapitel 2-6, können die folgenden Antworten auf die Fragestellungen sowie Empfehlungen für weiterführende Forschungsansätze gegeben werden.

I Wie hoch sind die technischen Potenziale für die Erzeugung von Bioenergie auf degradierten und marginalen Flächen, wenn unterschiedliche Rahmenbedingungen und verschiedene geografische Ebenen betrachtet werden?

Das technische Potenzial von Bioenergieerzeugung auf degradierten und marginalen Flächen wurde in dieser Arbeit für unterschiedliche Rahmenbedingungen und geografischen Ebenen ermittelt. Tabelle 7.2 gibt einen Überblick über die gefundenen Potenziale. Die Ergebnisse zeigen, dass ein technisches Potenzial von etwa 56 EJ y^{-1} für Bioenergieerzeugung auf versalzenen Flächen weltweit besteht (Tabelle 7.2). Nijsen et al. (submitted) schätzen das Bioenergieproduktionpotenzial von anthropogen degradierten Flächen (ohne versalzene Flächen) auf 32 EJ y^{-1} (siehe auch Abschnitt 1.2.1). Kombiniert man diesen Wert mit dem technischen Potenzial aus dieser Arbeit, dann kommt man zu dem Ergebnis, dass das technische Potenzial von Bioenergieerzeugung auf natürlichen und anthropogen degradierten und marginalen Flächen ca. 90 EJ y^{-1} beträgt. Dies entspricht etwa 18% der ca. 514 EJ y^{-1} des heutigen globalen Primärenergieverbrauchs (IEA, 2010). Ein Vergleich mit Ergebnissen anderer Studien zeigt, dass dieses Potenzial zwischen der Schätzung von Dornburg et al. (2010, 70 EJ y^{-1}) und dem oberen Ende der Spanne von Hoogwijk et al. (2003, 8-110 EJ y^{-1}) und Schubert et al. (2009, 34-120 EJ y^{-1}) liegt. Würde auch das Bioenergieproduktionspotenzial auf nicht-versalzenen ariden und semi-ariden Regionen in Subsahara-Afrika und anderswo berücksichtigt werden, könnte das zu einer weiteren Steigerung dieses Potenzials führen.

Tabelle 7.2: Übersicht über die technischen Bioenergiepotenziale von degradierten und marginalen Flächen unter Berücksichtigung unterschiedlicher Rahmenbedingungen und geografischer Ebenen sowie einer nachhaltigen Landnutzung

Kapitel	**Rahmenbedingungen**	**Geographische Ebenen**	**Umfang**		**Durchschnittsertrag**		**Potenzial**	
			Mha		t ha^{-1} y^{-1} [a]		EJ y^{-1}	
3	- Palmölproduktion auf *Imperata* Grasland	Malaysia	1		4,3	6,1[b]	0,1	0,2[b]
		Indonesia	12		3,5	5,9[b]	1,1	1,9[b]
4	- Produktion von Maniok-Ethanol, Jatropha-Öl und Brennholz auf marginalen semi-ariden und ariden Flächen	Botswana	8	4[c]	5,5	0,7[c]	0,8	0,1[c]
		Burkina Faso	2	0	10,0	6,3	0,3	1,0
		Kenya	3	3	12,4	8,9	0,7	0,5
		Mali	3	6	8,1	2,7	0,5	0,5
		Senegal	1	1	7,4	5,7	0,1	0,1
		Südafrika	1	0	8,7	1,1	0,1	0
		Tansania	2	0	12,4	n.a.	0,5	0
		Sambia	2	0	9,5	n.a.	0,3	0
		Gesamt	22	13	7,2	4,1	3,2	1,1
5	- Erzeugung von holzartiger Biomasse aus forstwirtschaftlichen Plantagen auf versalzenen Flächen	Global	971		3,1		56,2[d]	
		Regionen mit dem größten Potenzial:						
		Oceanien	144		7,6		20,2	
		ehemahlige UdSSR	117		4,7		10,0	
		Südamerika	57		5,2		5,4	
6	- Erzeugung von holzartiger Biomasse und Bioenergie aus (agro-)forstwirtschaftlichen Plantagen auf versalzenen Flächen	Regional: Gesamt Bangladesch, Indien und Pakistan	11		6,0		1,2	

a - Durchschnittsertrag wird für Palmöl in Tonnen rohes Palmöl (CPO) pro Hektar und Jahr und für Brennholz/holzartige Biomasse in Tonnen Trockenmasse pro Hektar und Jahr angegeben.

b - Erträge sind hier für das Jahr 2020 für das Basisszenario (base case; linke Spalte) und ein ertragoptimiertes Szenario (improved case; rechte Spalte) angegeben.
c - Umfang, Durchschnittserträge und Potenziale semi-arider (linke Spalte) und arider (rechte Spalte) Gebiete sind nur für Brennholz dargestellt. Erträge und Potenziale von Maniok-Ethanol und Jatropha-Öl werden in Tabelle 4.1 und Tabelle 4.4 (Kapitel 4) aufgeführt.
d - Umfang, Durchschnittserträge, und Potenziale von versalzenen Flächen sind für den Fall dargestellt, dass landwirtschaftliche Nutzflächen zur Verfügung stehen. Die Ergebnisse in dem Fall, dass wegen Nachhaltigkeitsanforderungen landwirtschaftlichen Nutzflächen ausgeschlossen werden, sind in Kapitel 5 dargestellt.
n.a. - Nicht zutreffend

Allerdings gibt es auch Faktoren, die nur in begrenztem Umfang in die Beurteilung des Bioenergieproduktionspotenzials von degradierten und marginalen Flächen einfließen. Die beiden wichtigsten Faktoren sind dabei die mögliche Verschärfung der Wasserknappheit in wasserarmen Regionen und die derzeitige Nutzung und Funktion von degradierten und marginalen Flächen. Beide Faktoren könnten das Potenzial für die nachhaltige Erzeugung von Bioenergie verringern, allerdings ist die tatsächliche Auswirkung dieser Faktoren auf das nachhaltige Potenzial derzeit unklar.

Im Hinblick auf eine mögliche Verschärfung der Wasserknappheit ist mehr Forschung zu den hydrologischen Auswirkungen der (großskaligen) Bioenergieproduktion auf verschiedenen geografischen Ebenen erforderlich, ebenso zu den wirtschaftlichen, sozialen und ökologischen Auswirkungen des verschärften Wettbewerbs um Wasser, zu den möglichen Auswirkungen der Produktion mehrjähriger Energiepflanzen auf die Infiltration und Wasserretention des Bodens und zur Art und Weise, wie diese Aspekte das nachhaltige Potenzial der Bioenergieproduktion auf degradierten und marginalen Flächen beeinflussen. Eine umfassende Beurteilung erfordert räumlich explizite Datensätze zu Grundwassertiefe und -qualität. Doch vor allem auf globaler und kontinentaler Ebene existieren derartige Datensätze nicht, wie die Beurteilung der globalen technischen und wirtschaftlichen Potenziale der Bioenergieproduktion auf versalzenen Flächen (siehe Kapitel 5) gezeigt hat. Die Forschungsanstrengungen sollten sich daher zunächst auf die Erhebung von mehr Grundwasserdaten auf nationaler Ebene konzentrieren, so dass auch detaillierte kontinentale und globale Datensätze zur Grundwassertiefe und -qualität erstellt werden können.

Die vorliegende Arbeit versucht, soweit wie möglich, Flächen für die Produktion von Bioenergie auszuschließen, bei denen wesentlichen Nutzungskonflikte existieren (z. B. Flächen, die für die Lebens- und Futtermittelproduktion genutzt werden oder Gebiete mit Bedeutung für die biologischer Vielfalt). Diesbezüglich existieren Unsicherheiten, weil die genutzten Datensätze nicht alle Nutzungsarten und Funktionen von potenziell geeigneten Flächen bewerten. Beispiele für Nutzungen und Funktionen, die besonders schwer zu erfassen sind, sind beispielsweise die Nutzung von Flächen für Weidevieh, die Nutzung durch Jäger und Sammler sowie die Ökosystem- und kulturellen (Dienst-)Leistungen von degradierten und marginalen Flächen. Werden diese Nutzungs weisen und Funktionen nicht in der Analyse berücksichtigt, so kann Bioenergieproduktion dazu führen, dass diese Nutzungen und Funktionen verdrängt oder dass die Potenziale zu hoch eingeschätzt werden. Zukünftige Forschung sollte daher insbesondere untersuchen, in wieweit spezifische Nutzungsformen und -funktionen die Verfügbarkeit von degradierten und marginalen Flächen für die Produktion von Bioenergie beeinflussen. Darüber hinaus muss bei der Planung oder der Untersuchung konkreter Bioenergieprojekte, eine Bewertung der heutigen Flächennutzung, Landbesitzverhältnisse und Funktionen durchgeführt werden, um nicht-nachhaltige Landnutzung, Verlust von Ökosystemfunktionen und negative soziale und ökologischen Folgen der Erzeugung von Bioenergie zu minimieren.

II Ist die Erzeugung von Bioenergie auf verschiedenen degradierten und marginalen Flächen, unter Berücksichtigung positiver Nebeneffekte, wirtschaftlich?

Die Ergebnisse dieser Arbeit zeigen, dass Biomasse- und Bioenergieproduktion auf degradierten und marginalen Flächen in verschiedenen Regionen wirtschaftlich sein können und darüber hinaus zur Deckung des lokalen und regionalen Energiebedarfs beitragen können. Kapitel 6 analysiert die Wirtschaftlichkeit von (agro-)forstwirtschaftlichen Systemen auf versalzenen Böden in Südasien im Hinblick auf den Kapitalwert. Obwohl der NPV in den drei Fallstudien positiv ist, zeigte unsere Analyse, dass die Wirtschaftlichkeit stark von den folgenden Faktoren abhängt: 1) Art und Schweregrad der Versalzung: beides beeinflusst die Art des (Agro-)Forstwirtschaftssystems und die gepflanzten Baumarten sowie den Energieertrag, 2) Umtriebszeit, 3) Märkte und Entwicklung der Marktpreise für Produkte aus Holz, 4) die Möglichkeit mit Emissionszertifikaten zu handeln und 5) Diskontsatz. Berücksichtigt man den ökonomischen Wert der Kohlenstoffspeicherung durch (agro-)forstwirtschaftliche Systeme, kann sich der Kapitalwert je nach Emissionsgutschrift (1-15 € t^{-1} CO_2-eq.) bis zu 129% erhöhen. Ein solches Szenario ist allerdings nur plausibel für den Fall, dass Agro-Forstwirtschaftssysteme für den Emissionshandel berechtigt sind. Teilnahmebedingungen und praktische Hürden für Kleinbauern, die sich an dem Emissionshandel beteiligen wollen, erfordern weitere Aufmerksamkeit.

Die Bioenergieproduktionskosten der verschiedenen Systeme auf degradierten und marginalen Flächen erwiesen sich als konkurrenzfähig mit den Marktpreisen relevanter Referenzsysteme. Beispiele hierfür sind die Produktion von Brennholz in vielen semi-ariden Regionen in Afrika südlich der Sahara, Jatropha-Öl und Maniok-Ethanolproduktion in semi-ariden Gebieten südlich der Sahara (Kapitel 4), und Brennholz- und Holzkohleproduktion in (agro-)forstwirtschaftlichen Plantagen auf versalzenen Böden in Südasien (Kapitel 6). Allerdings liegen die Kosten von Brennholz, Jatropha-Öl, und Maniok-Ethanolproduktion in den meisten ariden Gebieten Subsahara-Afrikas oft über den durchschnittlichen nationalen Marktpreisen für Brennholz, Diesel, und Benzin (Kapitel 4). Trotz der hohen Produktionskosten kann eine nachhaltige Erzeugung von Bioenergie in diesen Regionen wünschenswert sein, da dies potentiell die wirtschaftliche und soziale Entwicklung in ländlichen Gebieten fördert. Mögliche Nebeneffekte beinhalten zum Beispiel eine verstärkte lokale Energieproduktion und -versorgung, die Schaffung neuer

Märkte für landwirtschaftliche Produkte sowie die Erschließung alternativer Einkommensquellen für die ländliche Bevölkerung.

Anhand der Produktionskosten wurden dann Angebotskurven der Produktionskosten erstellt, um das wirtschaftliche Potenzial der Bioenergieproduktion auf degradierten und marginalen Flächen zu bestimmen (siehe Abbildung 4.4 und Abbildung 5.4). Diese Arbeit zeigte, dass das wirtschaftliche Potenzial für viele Regionen signifikant sein kann. Zum Beispiel schätzt Kapitel 4, dass das wirtschaftliche Potenzial der Brennholzproduktion in semi-ariden Gebieten in Tansania bei Produktionskosten von bis zu 2 € GJ^{-1} etwa 103 PJ y^{-1} beträgt. Dies entspricht 16% des heutigen Primärenergiebedarfs in Tansania. Kapitel 5 zeigt, dass die Erzeugung von Bioenergie auf versalzenen Böden zur Energieversorgung, insbesondere in Afrika, beitragen kann. Das gesamte wirtschaftliche Potenzial in Afrika beträgt bei Produktionskosten von bis zu 2 € GJ^{-1} etwa 8 EJ y^{-1}; dies entspricht etwa 28% des heutigen gesamten Primärenergieverbrauchs in Afrika. Das globale wirtschaftliche Potenzial der Biomasseproduktion auf versalzenen Böden (einschließlich landwirtschaftlich genutzter Flächen) beträgt bei Produktionskosten von bis zu 2 € GJ^{-1} rund 21 EJ y^{-1}, d.h. etwa 4% des weltweiten Primärenergieverbrauchs. Das wirtschaftliche Potenzial erhöht sich auf 53 EJ y^{-1}, wenn Biomasse mit Produktionskosten von bis zu 5 € GJ^{-1} berücksichtigt wird. Wenn landwirtschaftliche Flächen aus Gründen der Nachhaltigkeit ausgeschlossen werden, verringert sich das wirtschaftliche Potenzial der globalen biosalinen Forstwirtschaft: 1) auf 12 EJ y^{-1} bei Produktionskosten von bis zu 2 € GJ^{-1} und 2) auf 39 EJ y^{-1} bei Produktionskosten von bis zu 5 € GJ^{-1}.

III Welche ökologischen Auswirkungen hat die Bioenergieproduktion, unter Annahme verschiedener Rahmenbedingungen, auf degradierten und marginalen Flächen?

Die vorliegende Disseration zeigt, dass die Erzeugung von Bioenergie durch den Anbau von holzartiger Biomasse auf degradierten und marginalen Flächen sowohl positive als auch negative Umweltauswirkungen hat. Mögliche positive Auswirkungen entstehen durch die Speicherung von Kohlenstoff, die Erhöhung der Bodenfruchtbarkeit, eine potenzielle Verbesserung der Infiltration und Wasserretention des Bodens, eine Verringerung der Bodenerosion sowie eine Reduzierung der Bodenversalzung und -sodifzierung. Die detaillierte Bewertung der Treibhausgasbilanz am Beispiel der Bioenergieerzeugung aus Palmöl, welches auf *Imperata* Grassland in Malaysia angebaut wird (Kapitel 2) sowie am Beispiel der Biomasseproduktion durch (agro-)forstwirtschaftliche Systeme auf versalzenen Böden in Südasien (Kapitel 6) zeigt, dass die Erzeugung von Bioenergie auf degradierten und marginalen Flächen die Emissionen um mehr als 100% im Vergleich zu fossilen Brennstoffen verringern kann. Die Ergebnisse zeigen, dass Bioenergieplantagen auf degradierten und marginalen Flächen zu Kohlenstoffsenken werden können. Kapitel 2 ergab außerdem, dass Landnutzungs-änderungen der entscheidende Faktor in der Treibhausgasbilanz von Palmölenergie ist und dass, wenn ehemaliger natürlicher Regenwald oder Feuchtgebiete anstelle von degradierten Flächen zu Palmölproduktion genutzt werden, die Emissionsreduktionsziele der EU-Richtlinie Erneuerbare Energien nicht erfüllt werden können. Wenn allerdings Palmöl auf degradierten Flächen erzeugt wird und darüber hinaus das Plantagen-management verbessert wird, können die Emissionsreduktionsziele erfüllt werden und Palmöl-Strom in puncto THG-Emissionen als nachhaltig betrachtet werden.

Trotz dieser möglichen positiven Auswirkungen kann Bioenergieproduktion auf degradierten und marginalen Flächen auch Umweltrisiken für Böden, Biodiversität und Wasser darstellen. Zum Beispiel zeigte die Bewertung der (agro-)forstwirtschaftlichen Systeme auf versalzenen Böden in Südasien (Kapitel 6), dass sich salz-tolerante Baumarten, die auf degradierten und marginalen Flächen angebaut werden, auf nicht degradierten/marginal Flächen sowie in natürlichen Ökosystemen verbreiten können. Es wird daher vorgeschlagen, dass vor allem einheimische Baumarten gepflanzt werden und artspezifisches Management angewandt wird, um dieses Risiko zu minimieren. Ein weiteres Risiko besteht in der möglichen Verschärfung der Wasserknappheit in bereits wasserarmen Regionen. Wie bereits in der Antwort auf Forschungsfrage 1 angedeutet, ist die weiterführende Erforschung des Wasserbedarf einer groβskaligen Bioenergieproduktion auf degradierten und marginalen Flächen sowie die damit verbundenen wirtschaftlichen, sozialen und ökologischen (insbesondere hydrologischen) Auswirkungen erforderlich. Außerdem ist eine systematische Erforschung aller Auswirkungen erforderlich, die eine groβskalige Erzeugung von Bioenergie auf die natürliche Artenvielfalt sowie die komplexen ökologischen Prozesse in natürlichen Ökosystemen haben kann, um negative Auswirkungen zu minimieren. Darüber hinaus betonte Kapitel 4, dass die Erhebung detaillierter nationaler Datensätze über die biologische Vielfalt wichtig ist, um diese Informationen besser in die Beurteilung der Bioenergiepotenziale einzubeziehen.

Zusätzlich zu den Themen, die in dieser Arbeit behandelt wurden, müssen die sozialen und sozio-ökonomischen Auswirkungen der Bioenergieproduktion auf degradierten und marginalen Flächen untersucht werden, um deren Gesamtbeitrag zu einer nachhaltigen Entwicklung besser zu verstehen. Zwei wichtige Aspekte einer solchen Beurteilung sind: 1) die durch die Nutzung degradierter Flächen für die Bioenergieproduktion induzierten möglichen (in)direkten Landnutzungsänderungen und die damit verbundenen Chancen (zum Beispiel die Erwirtschaftung eines zusätzlichen Einkommens auf zuvor gering-produktiven Flächen) und Risiken (zum Beispiel die Verdrängung der Jäger/Sammeler und der nomadischen Hirtenvölker), und 2) die Bedingungen, in denen diese Risiken vermieden werden können.

Zusammenfassend können drei Hauptschlussfolgerungen gezogen werden. Erstens kann die Bioenergieproduktion auf degradierten und marginalen Flächen einen potenziell erheblichen Beitrag zur globalen Energieversorgung (ca. 18% des gegenwärtigen weltweiten Primärenergieverbrauchs) leisten. Zweitens suggeriert eine Analyse der Wirtschaftlichkeit, dass die Bioenergieerzeugung auf diesen Flächen in vielen Fällen im Wettbewerb mit anderen Bioenergieträgern und sogar mit fossilen Energieträgern stehen kann. Drittens kann die Kultivierung von mehrjährigen Energiepflanzen auf degradierten und marginalen Flächen sowohl positive (z. B. Kohlenstoffspeicherung, Verbesserung der Bodenfruchtbarkeit und Reduktion der Bodenerosion) als auch negative (z. B. Verschärfung der Wasserknappheit in wasserarmen Regionen) Umweltauswirkungen haben. Vertiefende Forschung ist notwendig, um die Bedingungen, unter denen Bioenergieproduktion zu negativen Auswirkungen auf die Umwelt führt, bestimmen zu können, so dass diese minimiert werden können. Aus diesen Ergebnissen lassen sich die folgenden zentralen Empfehlungen ableiten:

1) Demonstrationsprojekte unter verschiedenen Rahmenbedingungen sollten umgesetzt werden, um Erfahrungen bei der Errichtung und dem Betrieb einer nachhaltigen Bioenergieproduktion auf degradierten und marginalen Flächen zu sammeln.
2) Die Entwicklung einer Infrastruktur und der weitere Aufbau von Kapazitäten zur Bioenergieproduktion sind erforderlich. In Bezug auf den Aufbau von Kapazitäten ist dies besonders für die Errichtung und den Betrieb einer nachhaltigen Bioenergieproduktion auf unterschiedlichen Arten von degradierten und marginalen Flächen und im Hinblick auf die Minimierung von möglichen Umweltrisiken notwendig.
3) Die möglichen positiven Nebeneffekte der Bioenergieproduktion - wie zum Beispiel die Möglichkeiten zur Wiederherstellung degradierter Flächen, die Erwirtschaftung von zusätzlichem Einkommen auf bisher geringproduktiven oder unproduktiven Flächen, die Kohlenstoffspeicherung, die verbesserte Infiltration und Wasserretention des Bodens und die Verringerung von Bodenerosion - sind weitere wichtige Gründe für die Erforschung von und Investitionen in Bioenergieproduktion auf degradierten und marginalen Flächen. Bezieht man den ökonomischen Wert dieser Nebeneffekte in die Wirtschaftlichkeitsanalysen ein, so steigert sich die Attraktivität der Bioenergieproduktion auf degradierten und marginalen Flächen. Für die Internalisierung dieser externen Effekte sind (ökonomische) Anreize sowie eine effektive Umweltgesetzgebung notwendig.
4) Die Bioenergieproduktion auf degradierten und marginalen Flächen hat viele positive aber auch einige potenziell negative Auswirkungen auf die Umwelt. Eine umfassende Bewertung aller ökologischen, ökonomischen und sozialen Auswirkungen kann durch eine Bewertung mit Hilfe von Nachhaltigkeitskriterien im Rahmen einer offiziellen Zertifizierung erfolgen. Dadurch würden verifizierbare Anforderungen an die Erzeugung von Bioenergie auf degradierten und marginalen Flächen definiert und eine nachhaltige Flächenbewirtschaftung sichergestellt.

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[1] The only footnote of this thesis is dedicated to those people that contributed to this thesis but that, in the busy time of finalizing this book, I accidently forgot to mention by name. Please accept my sincere apologies and rest assured that I still appreciate your help.

Curriculum Vitae

Birka Wicke was born on September 29, 1979, in Kassel, Germany. She studied mathematics and environmental science at the University of North Carolina at Asheville (1999-2001). In 2001 and 2002 she travelled and worked in Spain before returning to the US to finish her bachelor's degree with honours in mathematics and environmental science at Tulane University (New Orleans, LA; 2002-2004). Throughout her bachelor's studies, Birka ran track/cross-country and received athletic scholarships. Later in 2004, she started a master's programme in sustainable development (energy and resources track) at Utrecht University in the Netherlands. Her M.Sc. thesis dealt with the macroeconomic impacts of bioenergy production and included field research in Argentina. Birka graduated with honours in 2006 and subsequently began work as a junior researcher in the Science, Technology and Society (STS) group in the Copernicus Institute (Faculty of Science) at Utrecht University. She worked on several different projects, the results of which are presented in this thesis. She presented her work at various scientific conferences and workshops in the Netherlands, Germany, Spain, Malaysia, India, and Burkina Faso. In addition to her research work, Birka was a teaching assistant in several courses related to energy and supervised M.Sc. thesis projects. After completing her doctoral work, Birka will be employed as a postdoctoral researcher at STS.

Peer-reviewed articles:

Wicke, B., Dornburg, V., Junginger, M., Faaij, A., 2008. Different palm oil production systems for energy purposes and their greenhouse gas implications. Biomass and Bioenergy 32(12): 1322-37.

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Wicke, B., Smeets, E., Watson, H., Faaij, A. The current bioenergy production potential of semi-arid and arid regions in sub-Sahara Africa. Accepted for publication in Biomass and Bioenergy.

(2 more articles are forthcoming, see Chapters 5 and 6)