Sustainable Development in the 21st century (SD21)

Sustainable land use for the 21st century

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Executive Summary

Introduction

It is estimated that the human footprint has affected 83% of the global terrestrial land surface and has degraded about 60% of the ecosystems services in the past 50 years alone. Land use and land cover (LUCC) change has been the most visible indicator of the human footprint and the most important driver of loss of biodiversity and other forms of land degradation.

Recent trends on global demand for food and bioenergy change – which are closely linked to food and energy price spikes and volatility – have raised concerns on the impact of LUCC change on biodiversity and other environmental impacts. Additionally, LUCC change could lead to natural resource degradation – which affect the poor the most since they heavily depend on natural resources. Since the earth Summit in 1992, the international community, individual countries, communities, civil society and businesses have increasingly become aware of the environmental impact of LUCC change. This paper assesses the LUCC change and explores factors which could be addressed to ensure sustainable development. The paper is divided into five sections and the first one begins by exploring what science tells us about LUCC change. The second section uses three case study countries to discuss how LUCC changes happen in practice. This is followed by an analysis of the land management programs and the effectiveness of market-based instruments. The fourth section discusses LUCC modeling and the last section concludes the paper by looking at the future prospects of LUCC change.

What does science tell us about LUCC change?

We explore what science tells us about LUCC using the major terrestrial land use types, namely forests and agriculture. We also discuss biodiversity and water resources, both of which are closely linked to each other and to agriculture and forest. Forest trends across regions are driven by economic development, government policies and other socio-economic factors. In the past two decades (1990-2010), forest density has increased globally while forest extent has slightly decreased by 0.2% per year in 1990-2000 and by 0.1% in 2000-10. Overall, forest density and extent has increased in high income countries and generally declined in low income countries. Related to forest is biodiversity, which is enhanced by establishment of protected areas. Globally, protected area increased by 38% in 2010 from its level in 1992. Despite the impressive increase in protected area, loss of biodiversity remains quite high since biodiversity is naturally developed over a long time and therefore increase in protected area is not matched with immediate increase in biodiversity – at least in the short-run.

An environmental Kuznets curve – which shows a decline in forest extent as the economy grows and subsequently an increase after reaching a threshold – explains LUCC change trends in most countries. Forest extent and density and biodiversity also reveal an environmental Kuznets curve pattern. Many countries in the tropics are in phase two – increasing forest extent and density and biodiversity. However, the forest transition – the environmental Kuznets curve of forest extent – has not been observed in some countries due to a number of reasons including strong timber markets, civil wars, government policies, etc. Additionally, the predictive power of the environmental Kuznets curve has been reduced by globalization and the increased role of international trade.

Rates of agricultural expansion are decreasing globally but still expanding in sub-Saharan Africa (SSA) and Latin American countries. The decrease in expansion of agricultural land is largely due to increasing agricultural productivity. From 1961-2005, crop yield accounted for 77% of the global increase in food production but in SSA, contribution of crop yield to total production was only 38%, the lowest among all regions. Hence the yield gap – the difference between potential and actual yield – remains wide in SSA and other developing countries south and central Asia but has narrowed in high income countries. It is in low income countries that there remains a large potential for increasing food production without increasing agricultural area. This requires investment to address constraints which contribute to low agricultural productivity, which include poor market infrastructure and generally low investment in agriculture.

Demand for water is increasing fast. It estimated that the growing human population will require more food, which will translate into doubling demand for water for agriculture from the current level of 7,130 km^3 to 12,050 to 13,500 km^3 in 2050. Additionally, water availability is expected to be less reliable in arid and semi-arid areas due to climate change, which is expected to increase intra-annual variability in precipitation and increased severity and frequency of droughts.

How does land use change happen in practice, and how are competing demands on land managed?

Brazil, DRC and Indonesia are used as case studies to illustrate the impact of country policies on LUCC change. Brazil – home to the largest part of the Amazon – has implemented policies which successively led to degradation and later rehabilitation of the Amazon. Until 2011, about 19% (762,000 km^2) of the original Amazon forest area was cleared under policies which encouraged colonization of the Amazon. About 72% of the forest clearing took place from 1980-2011. However, Federal, state, and municipal governments in Brazil realized the negative impacts of the losses and took actions to stop deforestation. In collaboration with international donors, Brazil was able to reduce the deforestation rate by 74% in only five years (2004-2009).

Similarly, Indonesia provided timber concessions which led to rapid deforestation. Agricultural expansion – especially palm oil production – and decentralization of forest management also contributed to deforestation. Decentralization of forest management coupled with limited local government budgets led local governments to use timber concessions to generate revenue. As in the case of Brazil, the Indonesian government – in collaboration with international donors – embarked on efforts to reduce deforestation. These efforts included strict enforcement of protected areas and incentives for protecting forest areas. Community forest management programs were also implemented. Recent data show that the annual deforestation rate in Indonesia fell from 1.7% in 1990-2000 to only 0.5% in 2000-10.

Forest trends in the Democratic Republic of Congo (DRC) show a significantly different pattern. Deforestation in DRC has been limited by the poor infrastructure and by insecurity, which deters commercial logging. However, there is a large informal logging activity run by chainsaw loggers and small-scale enterprises for domestic markets and illegal export to neighboring countries. Additionally, there has been a steady increase in legal concessions. Consequently the rate of deforestation has been increasing. With the help of the various international efforts, the country has recently taken several steps to protect the Congo forest and other natural resources. One of such efforts is the community forest. The decree establishing community forests is still pending and it would probably give large powers to the customary chiefs, who in DRC have often shown a limited sense of accountability. Overall, the reach of the public authorities is very limited in a country that remains a "fragile State", where corruption is still omnipresent and the judiciary system is down.

What do we know about the effectiveness of land management systems at the sectoral level?

There is increasing debate on the role of market-based instruments (MBIs) to reduce land use conversions and other environmental issues. MBIs have the potential to serve as an efficient alternative to administrative regulations and prescriptive laws for addressing environmental issues. Forest certification and eco-labeling have been probably the most successful MBI over the last two decades for enhancing sustainable forest harvesting and management. Forest products sold in high income countries with strict environmental standards are required to have a forest certificate showing that the products were not obtained from protected areas or other ecologically important areas. However, effectiveness of forest certification is largely restricted to a handful of companies exporting their products to environmentally-concerned markets.

In high income countries, conservation easement programs – bilateral contracting with land owners or users to not use land for certain development or use – have also shown considerable success. Other MBIs have been used but they have been more successful in high income countries than in low income countries.

Programs which have also been fairly successful in medium and low income countries are those aimed at enhancing Payments for ecosystem services (PES). The PES programs pay land owners/users to conduct environmentally friendly initiatives or to give up destructive practices. Interest in PES has increased rapidly over the past two decades. Today there are more than 300 programs implemented worldwide, predominantly used to address biodiversity, watershed services, carbon sequestration and landscape beauty. Empirical evidence tends to indicate that water-related PES have been more effective than others, probably because the buyers are the direct beneficiaries of the service, unlike for biodiversity and carbon PES where the beneficiaries are the global community and buyers are those serving as intermediaries for the present and future world community.

A major problem of PES is the compensation based on the opportunity cost. This has been regarded as inequitable for the poorest populations. Freezing user rights such as clearing, hunting or even the prospect of working in a forestry company deprives people of opportunities to lift themselves out of poverty. Additionally, elite capture in PES has also been reported.

Additionality has also remained a major challenge of PES programs. So do the competing priorities of protected areas. For example, the debate on the REDD+ revolves around three distinct interest networks – those who give priority to carbon, those who are concerned about biodiversity, and those who defend the rights of local and "indigenous" populations. As a result of this and other challenges, most decisions and rules and regulations on the REDD+ funded by governments and international organizations are still pending. The weak prospects of climate change negotiations are posing another challenge to PES programs. In sum, PES and other MBI programs in general continue to face daunting challenges to reach agreement on a number of contentious issues. However the significant progress in international cooperation on sustainable development made in the past 20 years offers some hope and lessons for facing such challenges.

How are land use and land use modeled in scenario exercises?

Models for predicting future LUCC change use theory to link changes with its biophysical and socio-economic drivers. Statistical approaches are then used to establish historical relationship between LUCC and its drivers. Three main types of models have evolved based on different disciplines: geographic, economic and ecological models.

Geographic models are focused on land allocation based on suitability of land use and the spatial location of ecosystems and population. Hence geographic models tend to better allocate land use to areas with minimal effect on the ecosystems. The models better capture the potential productivity of different land uses and are better able to reflect land management than economic models. However, geographic models assume that prices and other international feedback variables are exogenous. This makes them less able to reflect the influence of international trade on market-driven agent behavior. *Economic models* focus on the demand and supply of land-based goods and services. They more effectively reflect the effect of international trade and globalization on LUCC change. Additionally, economic models use scenarios to capture the influence of policies and other socio-economic factors on LUCC. *Ecological models* link land allocation to species abundance and extinction, ecological footprints and other environmental concerns. Ecological modeling methods also often assume that prices and other economic variables are exogenous factors, thus failing to fully account for their impacts and associated trade-offs in land allocation.

Over time, LUCC modeling has become more integrated, breaking the disciplinary divide. In fact, the predictive accuracy of integrated models is higher than those of the specialized models. Such integrated approach fits well with the ecological interrelationships of different land uses and the integrated approach that characterizes sustainable development. For example, solutions to simultaneously achieve the food security, biodiversity and bioenergy objectives of maximizing human welfare require use of integrated models. Despite such progress however, prediction of future LUCC remains a challenge. Unforeseeable shocks and events as well as incorporation of human behavior in LUCC models have remained elusive and have contributed to poor prediction.

Prospects for the future

LUCC change is posing a grave danger to earth's ecosystems. One estimate puts the safe upper boundary for global cropland area to 15% of the total terrestrial area, a level that is only about three percentage point higher than current cropland area –which account for 12% of global land area. However, another estimate by the UNFCCC Commission has concluded that current global agricultural production has already stepped outside the safe boundary. Loss of biodiversity is already outside the upper boundary with the current rate of extinction being 100-1,000 times higher than the pre-industrial age level. Additionally, freshwater resources are also overwhelmed by the increasing population and climate change, which have increased their variability and reduced their supply in dry areas. The increasing demand for bioenergy has posed yet another challenge to land and water resources. Half of the global cereals consumption in 2005/6-2007/8 was due to US ethanol production and projections by FAO/OECD show that 52% of maize and 32% of oilseeds demand to year 2020 will be due to bioenergy. Estimates show that a large portion of the area for bioenergy production will be derived from clearing forests and grassland. These trends show that business as usual is not sustainable.

So what can be done to sustainably achieve food security, protect biodiversity and energy security? A recent forecasting study showed a decreasing yield growth at the global level. Food security is achievable but this will require increasing food production by increasing agricultural productivity in low income countries where the yield gap is widest. This will require addressing constraints which limit higher yield in such regions. These include increased investment in agricultural research as well as addressing market conditions and rural services, which will provide technical support and incentives for increasing productivity. Achieving food security also requires reducing post-harvest losses, which are high in both developing and developed countries. Post-harvest food losses could be reduced by investment in processing and storage investment in developing countries and by public awareness in developed countries to change food consumption habits which lead to food losses. Greater water productivity is also required to increase yield in the regions where water productivity is low.

On bioenergy, studies have cast doubt on the efficacy of biofuels as mechanisms for reducing GHG using current technologies. Efforts to use second generation feedstock provide some potential for liquid bioenergy which does not compromise food security and biodiversity.

Given the current high biodiversity losses, reducing biodiversity loss to pre-industrial levels will be hard to achieve. However, significant reduction in biodiversity loss is possible. In this regard, the recent increase in protected areas offers some hope.

Finally, prospects for international instruments for land use change management require synergistic programs, which provide several ecosystem services. This means international cooperation on carbon and other ecosystem service initiatives need to explore closer collaboration to achieve synergistic objectives. For example, closer collaboration of UNCCD, CBD, UNFCCC and others can simultaneously combat land degradation, conservation of biodiversity and carbon sequestration. This is in line with the spirit of Agenda 21, which promotes cooperation and the building of synergies among ecosystem initiatives.

Acronyms and abbreviations

CBD Convention on Biological Diversity
GHG Greenhouse gases
HANPP Human appropriation of net primary production
LAC Latin America and the Caribbean
LUCC Land use and land cover change
MA Millennium Ecosystem Assessment
PES Payment for Ecosystem Services
REDD Reduced emissions from avoided deforestation
SSA Sub-Saharan Africa
UNFCCC United Nations Framework Convention on Climate Change
UNCCD United Nations Convention to Combat Desertification

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Introduction

It is estimated that the human footprint has affected 83% of the global terrestrial land surface (Sanderson *et al* 2002) and has degraded about 60% of the ecosystems services in the past 50 years alone (MA 2005). The rates of land use and land cover change (LUCC), which had already increased in the last century, accelerated in the last three decades at an alarming level (Lambin and Geist 2006). These LUCCs mostly impacted humid and sub-humid areas (Bai *et al* 2008) that were largely along roads (Lambin and Geist 2006) and in agricultural areas. Today, agriculture occupies 38% of the globe's ice-free terrestrial surface and is the largest land cover type by area (FAOSTAT 2011a).

Since the 1992 Rio summit, the global community has become increasingly aware of the environment and the need for sustainable development through a range of practices and policies, including reduced deforestation, increased environmental monitoring, agricultural intensification, restoration of degraded landscapes, reduction of environmental pollution, and payment for environmental services (PES). Following the publication of the Millennium Ecosystem Assessment in 2005, sustainable development efforts have increasingly become more inclusive to focus on all components of ecosystems. According to Costanza *et al* (1997), ecosystems are goods and services provided by living organisms and their habitat with direct and indirect benefit to human populations. According to MA (2005), ecosystem goods and services include:

- **Provisioning services:** Goods provided — food, fiber, forage, fuelwood, pharmaceutical products, biochemicals, fresh water, etc.

- **Supporting services:** Services that maintain the conditions of life on Earth — soil development (conservation/formation), primary production, nutrient cycling

- **Regulating services:** Benefits obtained from the regulation of ecosystem processes — water regulation, pollination/seeds, climate regulation (local and global)

- **Cultural services:** Intangible benefits obtained from natural ecosystem — including recreation, landscapes, heritage, aesthetic, etc.

The economic value of supporting and regulating these services is not well-captured in the market and therefore always undervalued. Additionally, the benefit of the non-marketed ecosystem services is not yet well-known (Balmford *et al* 2002). Such a lack of knowledge and apparent lack of economic value poses a challenge for determining land use allocation and modeling land use change. However, what is known about the non-marketed ecosystem services suggests that they are likely to have a larger value than marketed ecosystems. For example, Constanza *et al* (1997) estimated the annual economic value of 17 ecosystem services – most of which were not traded in the market and therefore not considered in the traditional GDP and other economic statistics – to be about US$ 38 trillion per year (adjusted for the 2000 value). The equivalent GDP in the same year was US$32.216 trillion (IMF 2011).

This paper assesses LUCC since 1992 and explores related factors which should be addressed to ensure sustainable development. It starts by analyzing what science tells about LUCC, and then examines how LUCC happens in practice and how countries and the global community are managing the competing demand for land. The third section of the paper examines what we know about the effectiveness of different land management systems. The fourth section reviews the state of knowledge on LUCC models. The final section discusses prospects for the future and draws policy implications.

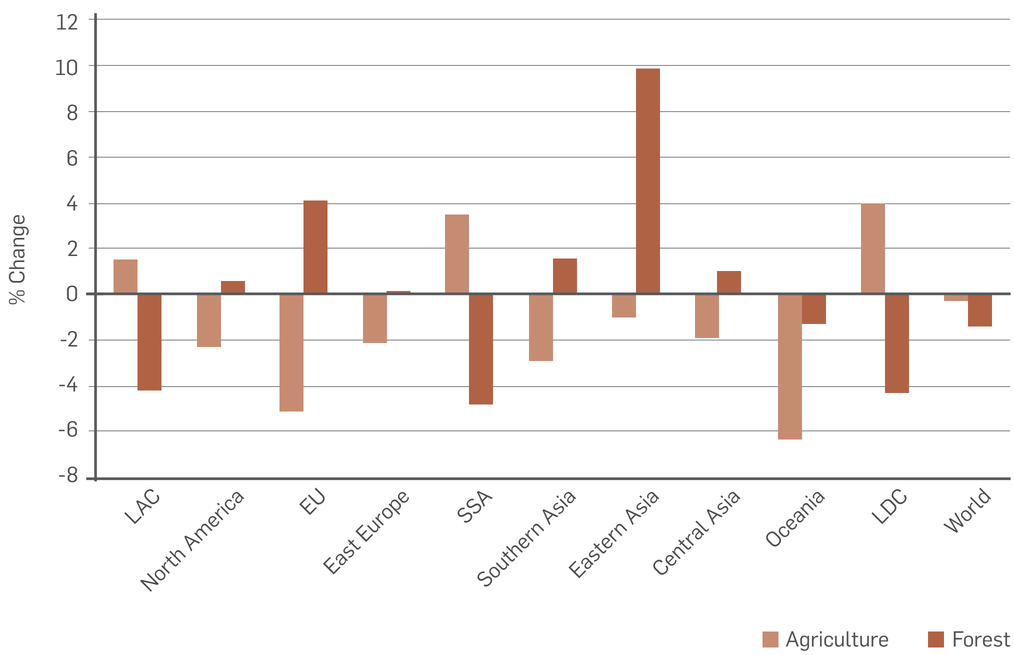
Science of land use and land cover change

We explore what science tells us about LUCC using the major terrestrial land use types, namely forests and agriculture. The two terrestrial ecosystems are closely linked to each other and to water and biodiversity. The relationships of the ecosystems are complex and this drives the increasing need to analyze and treat them as ecosystem services, the source of provisioning, cultural and regulating services that are crucial to human welfare (MA 2005; Rockström *et al* 2009; Constanza 2011). In effort to determine the trend of all ecosystem services, we examine the ecological footprint, a resource accounting tool that compares two opposing processes: 1) the biological capacity of land and sea area to produce food, fiber, timber, energy, absorb by products of consumption and provide space for infrastructure using the prevailing technology and 2) the demand for these ecosystem services by a given population (Kitzes and Wackernagel 2009).

Forests[1]

In 2010, the forest area covered 31% of global land area and was equivalent to 0.6 ha per capita (FAO 2010). Globally, deforestation, the permanent clearing of forests, decreased by almost 20% from 16 million ha year^{-1} in 1990-2000 to 13 million ha year^{-1} in 2000-2010 (FAO 2010). Sub-Saharan Africa (SSA) and South America contributed the largest share of deforested area in the last two decades (Figure 1). Recently, Brazil and Indonesia have significantly reduced deforestation rates. Australia saw an increase in forest loss, largely due to drought and forest fires (FAO 2010). The forest trends in Australia underscore the role played by biophysical factors in forest cover trends. As will be seen in the discussion of forest transitions and drivers of LUCC, forest trends across regions are driven by economic development, government policies and other socio-economic factors.

FIGURE 1 Change of agricultural and forest area, 1992-2009.



Note: Change computed as follows $\frac{y_2 - y_1}{y^1} * 100$, where y_1 = average area 1992-2000 and y_2 = average area 2001-2009.

LAC = Latin American Countries; SSA = sub-Saharan Africa; EU = European Union.

Source: FAOSTAT data.

Forest density, tree density per hectare, is the second factor of interest in forest change. In the past two decades (1990-2010), forest density has increased globally. The increase in forest density was most pronounced in North America and Europe; the increase in Africa and South America was only modest (Rautiainen *et al* 2011). Overall, forest density increased in 68 countries and accounted for 72% of the global forest area and 68% of global carbon mass (Rautiainen *et al* 2011). In Asia, forest density increased in 1990-2000 but decreased in 2000-2010 as forest area increased significantly, largely from afforestation in China. Conversely, deforestation rates and net losses in South and Southeast Asia increased (Rautiainen *et al* 2011).

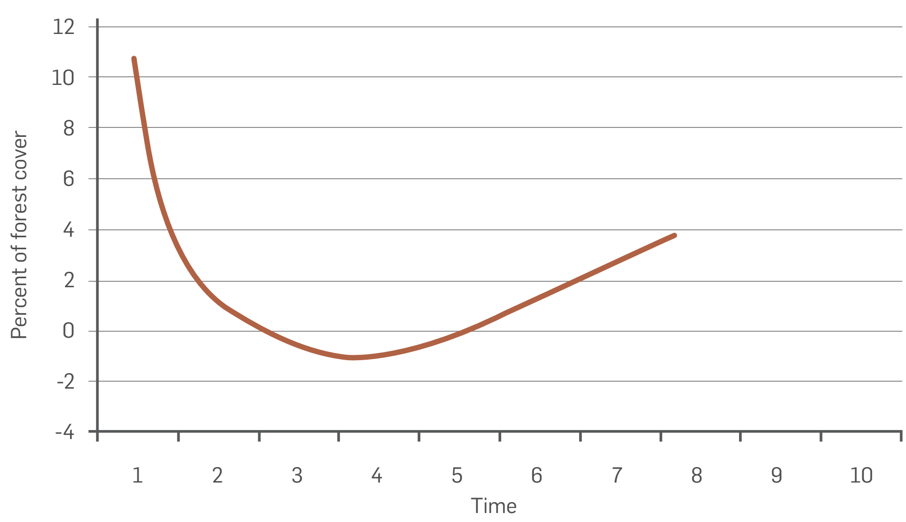
Forest transitions

Forest transition theory offers some explanation behind the trend of forest extent and density across countries and regions. As it will be seen below, this theory has been tested empirically and shown to be valid with some evidence showing different patterns (e.g. see Meyfroidt and Lambin 2011). Forest transition focuses on the forest stock change that has a predictable relationship with economic development (Mather 1990). Forest transition has three phases represented by the environmental Kuznets curve. In the first phase, increases in deforestation as the economy and population grow prompt greater demand for agricultural and forest products. In the second stage, migration to urban areas, increasing rural labor wage rates, and intensifying farming reduces labor demand. The value of forest products also increases. In the third phase, forest recovery begins when a threshold period is reached during which the value of forest products has increased, prompting land owners to protect and/or plant trees (Cooke *et al* 2008) (Figure 2).

In most of the tropics today, countries are in phase two, when urban population growth and agricultural exports are the primary drivers of deforestation (DeFries *et al*. 2010). For some countries (Ethiopia, Haiti and Togo), forest transition has not occurred largely due to a lack of alternative employment and/or institutions that could enhance tree planting (Ruddel *et al* 2005). Others (Burundi, El Salvador, Rwanda and Sierra Leone), have been embroiled in the insecurity of war, which has led to deforestation. In other countries (Brazil, Indonesia and Cameroon), strong forest product markets throughout the early 21st century caused deforestation despite significant economic development (Ruddel *et al* 2005). Recently, however, Brazil has reduced deforestation by more than two thirds in only five years due to aggressive policies and international cooperation. Increases in the country's forest plantations for paper, charcoal and chip board production have reduced pressure on pristine forests. In the 1990s, 38% of all countries experienced an increase in forest area after deforestation, suggesting that they reached the threshold and moved towards phase three (Ruddel *et al* 2005). Most European countries and all of North America experienced forest recovery in the 20th century largely due to general industrialization and economic development. Forest recovery in Asia, including the recovery in China, India and Bangladesh, exhibits a different pattern. Rural poverty in these countries remains entrenched, but the increasing value of forest products has spurred rural communities and the government to plant trees (Foster and Rosenzweig, 2003; Fang *et al* 2001). Government tree-planting programs, which were implemented to head off flooding, wind storms and other disasters attributable to deforestation, also helped to increase forest area (Ibid).

However, the predictive power of the forest transition model is being affected by globalization and the increased role of international trade. Similarly, government policies, such as the one implemented in Brazil to control deforestation at the municipal level through the creation of a black list of municipalities that most contribute to deforestation, could also change the forest transition by fastening the recovery process. For example, recent analysis suggests that the relationship between rural populations and forest cover has weakened as globalization has linked well-capitalized ranchers, farmers and loggers, and their products with distant markets (Rudel *et al*. 2009).

FIGURE 2 Forest transitions – stylized model.



Source: Authors' illustration.

A major conclusion from the forest transition analysis discussed above is that while economic development and forest product scarcity could trigger an increase in forest area, other socio-economic characteristics may inhibit forest recovery. As shown in Asia, government intervention could help forest recovery. In Niger for example, the government passed a statute (rural code) giving land owners tenure security of any tree that they plant or protect (Larwanou, Abdoulaye, and Reij 2006). It is estimated that at least 3 million hectares of land have been rehabilitated through tree protection, which allowed for natural regeneration (Adam *et al* 2006).

However, the rural code was not the only deciding factor that led to this remarkable success. The prolonged drought that spanned the 1970s and 1980s led to loss of trees, increasing the price of tree products. This provided strong incentive to farmers to plant and protects trees. Planted forest area as a share of total forest area in Niger was 12% in 2010 and was among the highest in SSA (FAO 2010). As discussed in Box 1, this achievement was a result of a combination of efforts by local communities, change in government policies and statutes, support from NGOs, and religious organizations and environmental stress, which prompted communities for a solution.

BOX 1 Regreening of the Sahel in Niger.

Regreening the Sahel in Niger is a success story due to its remarkable progress in planting and protecting trees that resulted from a combination of initiatives by the government, local communities, donors, NGOs, and religious organizations. Starting in the 1970s, in response to extensive vegetation loss due to droughts that lasted until the 1980s, the Nigerien government aggressively promoted tree protection and planting. One measure was recasting Independence Day as National Tree Day. Additionally, since the 1980s, more than 50 government programs – including the Special Program of the President and the Projet de Gestion des Ressources Naturelles (Natural Resources Management Program) – were promoted by the government, NGOs, and donors (World Bank 2009). NGOs and religious organizations involved in these efforts mobilized communities to plant and protect trees. They also built the capacity of local communities to manage natural resources. For example, a religious organization initiated the farmer-managed natural regeneration (FMNR)—in which communities protect or plant new trees and in return harvest fuelwood, fodder, nitrogen fixation from leguminous trees, windbreaks, and other ecosystem benefits (Reij, Tappan, and Smale 2008).

The government also revised its institutions and passed the rural code in 1993. This legislation gave customary leaders more land management power and encouraged them to plant and protect trees and to benefit from such efforts without government intervention. The forest policy gave landholders tenure rights to trees that they planted or protected (Yatich *et al*. 2008; World Bank 2009). The changes provided incentives for communities to plant and protect trees and helped them to cope with risky agricultural production.

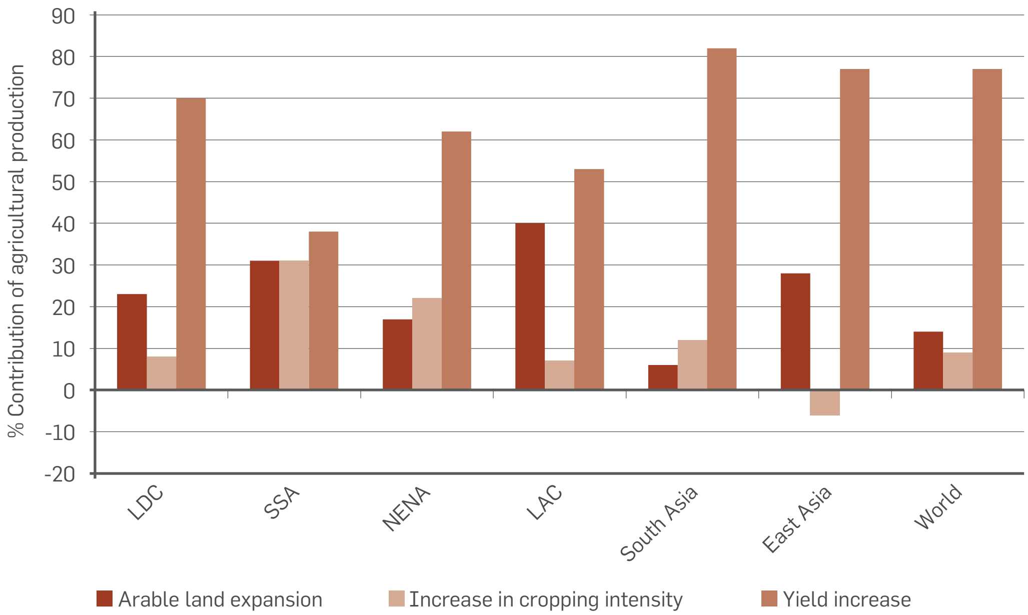
Additional policy changes and efforts by donors and NGOs also followed the 1970s–1980s drought, creating a new value for trees. Firewood and water as well as livestock were in short supply following the drought. The loss of livestock wiped out the traditional strategy of using livestock as buffer stock against shocks (Fafchamps *et al* 1998) – especially in northern Niger, where trees are used as fodder during the dry season. People responded to this challenge by protecting growing trees instead of cutting them, as had been the case in the past. Hence, tree scarcity significantly affected the livelihoods of rural communities, prompting them to change from land clearing to tree protection.

Studies carried out to understand the drivers of regreening of the Sahel found that villages where tree planting and protection projects were operating were much greener than what could be explained by change in rainfall (Herrmann, Anyamba, and Tucker 2005). It is estimated that villages with FMNR had 10–20 times more trees than they had had before FMNR started. Contrary to expectations, tree planting and natural regeneration in villages with higher population density were higher than in villages with lower population density (Reij, Tappan, and Smale 2008).

Agriculture

Agriculture is the leading form of human appropriation of net primary production (HANPP), which is human's harvest of photosynthetic products or alteration of photosynthetic production. HANPP influences biodiversity, reduces energy flow to non-human species and influences the provision of other ecosystems (Harbel *et al* 2007). Harbel *et al* (2007) estimate that even though cropland occupies only 12% of the global land area, crop harvest accounted for 49.8% of the HANPP in 2000 and grazing accounted for 28.9% of HANPP in the same year. Rates of agricultural expansion are decreasing globally but still expanding in SSA and Latin American countries (LAC) (Figure 1). It is estimated that 70% of the grasslands, 50% of the savanna, 45% of the temperate deciduous forests and 27% of the tropical forests have been cleared for agriculture (Foley *et al* 2011). Between 1992 and 2009, agricultural land area increased by about 4% in SSA-- the largest increase in all regions considered (Figure 1).

FIGURE 3 Source of growth of agriculture, 1961-2005.



Notes: LDC = Least Developed Countries; SSA = Sub-Saharan Africa; NENA = Near East and North Africa
Source: Computed from Bruinsma (2009).

About 80% of agricultural expansion in the tropics replaces forests, leading to serious consequences for biodiversity, greenhouse gas emissions and other environmental outcomes. Theory posits that population growth induces intensification (Boserup 1965) but such intensification occurs only if prices, markets and other socio-economic conditions favor such decision (Mortimore and Harris 2005; Boyd and Slaymaker 2000). If price and market conditions do not permit intensification, population density has been associated with land degradation (Grepperrud 1996; Scherr 2000).

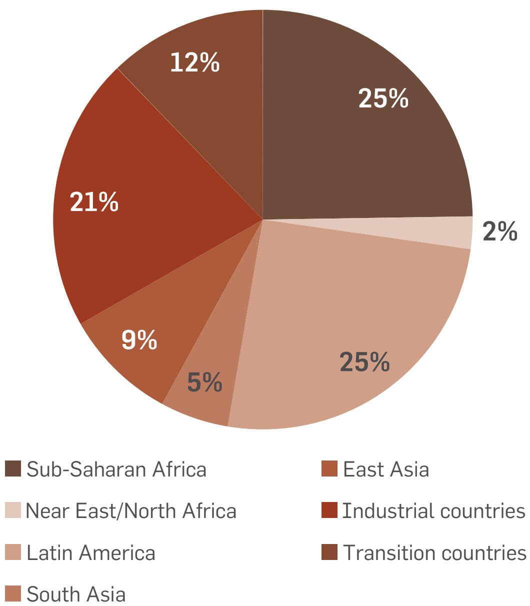
Agricultural intensification has occurred in all regions – including SSA. Figure 3 shows that from 1961-2005, crop yield accounted for 77% of the global increase in agricultural production but in SSA, contribution of crop yield to total production was only 38%, the lowest among all regions. The small contribution of yield increase to total production was due to the poor market conditions that provide incentives for farmers to invest more in increasing productivity. The global yield of crops increased by 47% between 1965-1985, but only increased by 20% between 1985-2005 (Foley *et al* 2011).

Where has agriculture been expanding?

Agriculture can only expand in an area that provides the ecological requirements of crops or livestock. FAO's Global Agro-Ecological Zone (GAEZ) defines suitable land as land with soil, terrain and climate characteristics which meet the crop production requirements with specified input levels (Fischer *et al.*, 2002). On a regional scale, the countries of Latin American and Caribbean (LAC) and SSA account for the largest share of arable land (Figure 4). However, the largest share of arable land has already been put under use and available land for expansion is limited. About 90% of the remaining 1.8 billion ha of arable land in developing countries is in LAC and SSA (Bruinsma 2009). Seven countries (Brazil, Democratic Republic of the Congo, Angola, Sudan, Argentina, Colombia and Bolivia) account for about 50% of the remaining suitable land (Ibid). Regions that have virtually run out of suitable land for expansion include South Asia and the NENA (Near East/North Africa). Expansion of agricultural land in such regions requires investment in irrigation or other soil amelioration measures.

It is also in countries with large arable land area that there are still large gaps between agricultural yield potential and actual yield. Such a large gap provides the potential for increasing agricultural production to cater to the increasing demand for agricultural products. As will be seen below, closing the wide agricultural productivity gap requires significant investment to address constraints which lead to the low agricultural productivity.

FIGURE 4 Contribution of regions to global suitable land.



Source: Bruinsma (2009).

Water resources

There is abundant supply of freshwater (47.97 million km^3) per year but only a small share is available at the right time and place. For example, only a third of the 110,000 km^3 annual precipitation reaches rivers, lakes and the aquifers, of which only 12,000 km^3 is available for irrigation, domestic and industrial use (SIWI *et al* 2005). Two thirds of precipitation is absorbed as soil moisture or evaporates (Ibid). Only 0.79% of freshwater is not frozen in ice or glaciers and of that, a large share is groundwater.

Irrigation water use has tripled in the past 50 years, and irrigation accounts for 70% of global freshwater withdrawals (UN water 2009). One estimate – among many estimates with different volumes – of current water use for food production is 6800 km^3/year (Shiklomanov, 2000). The world population growth of 80 million per year translates to an additional annual demand of freshwater of 64km^3 (UN water 2009). It is estimated that the growing human population will require more food, which will translate into doubling demand for water for agriculture from the current level of 7,130 km^3 to 12,050 to 13,500 km^3 in 2050 (CA 2007). According to another estimate, additional water withdrawal and use equivalent to 5,600 km^3/year would be required to eliminate hunger and undernourishment and to feed the additional three billion inhabitants in 2050 (Falkenmark and Rockström, 2004), which is about three times the water used for irrigation today (Shiklomanov, 2000).

Water scarcity is already evident in dry areas. Molden *et al* (2007) estimate that about 25% of the earth's river basins run dry before reaching the ocean due to water use. In 2000 about 2.3 billion people lived in river basins with water stress, i.e. had access to less than 1,700 m^3 per capita/year, below which, disruptive water shortages can frequently occur (Revenga, *et al* 2000). By 2025, Revenga *et al* (2000) estimated that 48% of the global population will have water stress under business as usual.

The irrigation water supply reliability index (IWSR) – a measure of availability of water relative to full water demand for irrigation – is also projected to decline from 0.71 globally in 2000 to 0.66 by 2050 (Ringler and Nkonya 2012). Additionally, water availability is expected to be less reliable in arid and semi-arid areas due to climate change, which is expected to increase intra-annual variability in precipitation and increased severity and frequency of droughts (Meehl *et al*. 2007). It is estimated that climate change will account for 20% of the increase in global water scarcity (UN 2003).

The increasing water scarcity calls for strategies to address the water stress – an aspect that could help increase agricultural productivity, which in turn will reduce conversion of natural ecosystems to agriculture. We analyze these strategies in the last chapter.

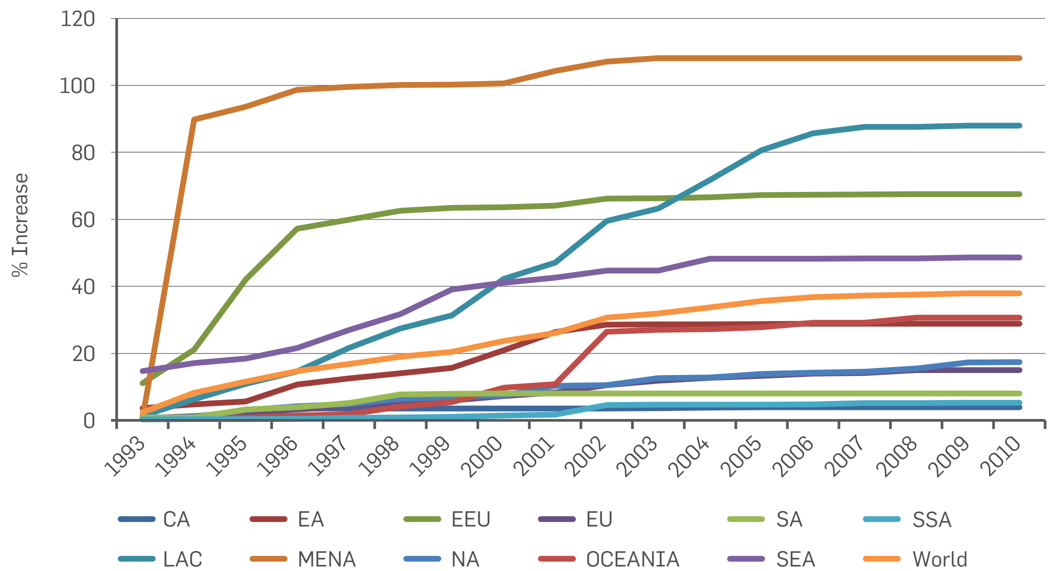
Biodiversity

Biodiversity provides a variety of ecosystem services, which, for a long time, have been ignored or undervalued. Greater biodiversity ensures more stable and resilient ecosystems. The Chennai Declaration states that biodiversity must be conserved because it is the raw material for food and health nutrition and provides material for biotechnology industry (Koohafkan and Altieri 2011). Hence changes in the abundance and diversity of species may have serious impacts on human welfare. For example, up to 80% of people from developing countries rely on wild flora and fauna for health care and wild meats provide 30-80% of protein for many rural communities (Nasi *et al* 2011).

Realizing the rapid loss of biodiversity and its potential impact on ecosystems and consequently human welfare, 193 of the 194 countries in the world are signatories of the CBD and 170 countries have already prepared their national strategies and action plans (NBSAPs). The CBD, ratified in 1992, set 11 goals[2] with 21 specific sub-targets to be achieved by 2010 (CBD 2010). A 2010 evaluation of the achievement of the sub-targets done in 2010 showed that while no single goal has been fully achieved; there has been significant reduction in the rate of biodiversity loss for most of the sub-targets (CBD 2010).

Since 1992, the protected area in all regions has increased significantly (Figure 5). Globally, the protected area increased by 38% from its level in 1992. The increase was especially large in the Middle East and North Africa (MENA) region, due to the fourfold increase by Saudi Arabia, whose of protected area increased about 4 times from about 148,000 km^2 in 1993 to 588,000 km^2 in 1994. The Latin and Caribbean countries (LAC) countries, which account for the largest share (25%) of global protected area, saw the second largest increase. This increase largely was due to Brazil, where protected areas increased almost threefold - from 812,000 km^2 in 1992 to 2.242 million km^2 in 2010. However ambitious plans to accelerate economic development in Brazil could threaten protected areas.

FIGURE 5 Change in protected area in global sub-regions from 1992 level.

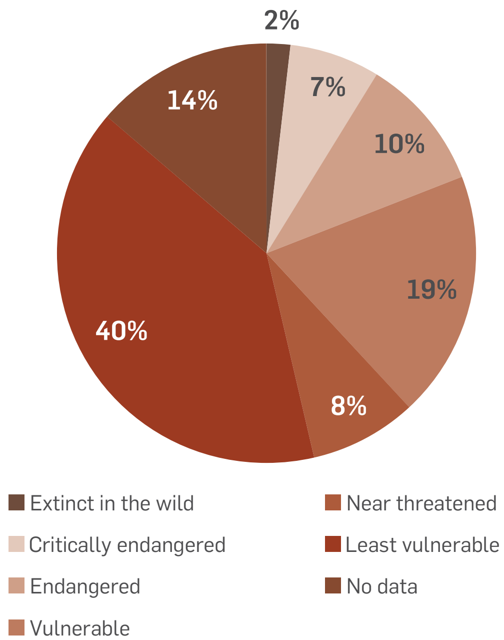


Notes: CA=Central Asia; EA=East Asia; EEU=East Europe; LAC=Latin America; MENA=Middle East and Northern Africa; NA=North America; SA=South Asia; SEA=Southeast Asia; SSA=sub-Saharan Africa.

Source: IUCN and UNEP-WCMC (2011).

Despite the impressive increase in protected area, loss of biodiversity remains quite high. The 2009 report of the Convention of Biological Diversity (CDB) shows that 36% of the 47,677 species already assessed are threatened by extinction (Figure 6), i.e., under the current trends, the species will become extinct (CBD 2010). Rockström *et al* (2009) also report that on average, more than 100 species extinction per million species per year (E/MSY) are lost, a level which is more than 100 times the planetary boundary (10 E/MSY) deemed to be earth system's safe operating space for human welfare. Current rate of extinction is 100-1000 higher than the Holocene (pre-industrial) age level (0.1 – 1 E/MSY) (Ibid).

FIGURE 6 Global biodiversity conditions.



Source: IUCN (cited by CBD 2010).

Even though recent efforts to increase extent of forest cover through reforestation and afforestation programs have helped to reduce net deforestation, they do not fully restore lost biodiversity since such natural biodiversity was built over hundreds of years and is composed of complex and diverse biomes. Species that have shown the most rapid decline include birds, mammals, and amphibians (hunted for food) and medicinal plants (Ibid). Terrestrial biodiversity losses have been driven by habitat loss and degradation through slash-and-burn clearing, forest fires, land use conversions, over-exploitation of plant and animal species, climate change, pollution, and invasive and alien species (CBD 2010).

Projections show that the impact of climate change on biodiversity loss is expected to increase in future. For example, the loss of ice sheets and the melting of permafrost in the arctic and Antarctic regions are threatening polar biomes.

There are underlying causes of biodiversity loss. For example, a study covering 73 countries examined the relationship between income and the threat of extinction for four major species (mammals, birds, plants and reptiles). It showed a U-shaped quadratic relationship (Perrings and Halkos 2010) – i.e., the Kuznets curve comparable with the forest transition pattern discussed above. The turning points differ for each country but the results were very robust (Perrings 2010).

Ecosystem services and ecological footprints

Ecosystem services are the "components of nature, directly enjoyed, consumed, or used to yield human well-being" (Boyd and Banzhaf 2006). The ecosystem services trend is analyzed using the concept of ecological footprint. The ecological footprint is a measure of the biologically productive land area and water required to provide food, feed, fiber, timber, energy and to absorb CO_2 waste under current technology (WWF 2012). The ecological footprint represents the demand for ecological services. The supply of ecological services is represented by biocapacity, which is the area of productive land and water available to produce resources or absorb CO_2, under current management practices. The net biocapacity is the difference between biocapacity and ecological footprint. Estimates show that since the mid-1980s, the world entered an ecological deficit, i.e., earth's biological capacity to produce ecosystem goods and services and human demand for provisioning services and regulating services – mainly absorption of CO_2 - was surpassed (GFN 2010). The ecological footprint is now estimated to be 45% higher than the earth's biocapacity, i.e., it takes the earth about 1.5 years to produce what is required by the global human population and to absorb the CO_2 produced using the current technologies (Ibid).

The ecological deficit is highest in North America and Europe, where carbon emissions account for the largest demand for land area to absorb CO_2. Oceania and LAC have the largest surplus ecological balance, while Africa has a delicate balance with only cropland having a slight negative balance of 0.06 global hectares (gha). Global population growth has been the largest factor contributing to the ecological deficit (GFN 2010). The population dynamics in SSA poses the biggest challenge since it is only in this region that population is expected to continue growing beyond 2100 and that the current growth rate is the highest (UNFPA 2011).

Main drivers of change and sources of pressure on land-use

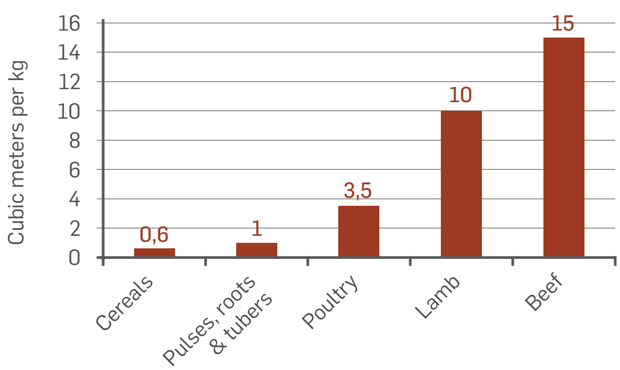
Population growth

Population has been the major driver of agricultural expansion (Foley *et al.*, 2011; Bruinsma, 2009; Ramankutty *et al.*, 2002). Even though the positive correlation between population growth and cropland is expected due to increasing demand for food, it is also true that people tend to settle in areas suitable for agriculture (Ibid). Technological development and international trade have weakened the relationship between population and expansion as well as settlement in areas of arable land (Ibid). For example, the Green Revolution in Asia led to a much slower area expansion than would have been the case without the productivity improvement. Borlaug (2000) estimates that if cereal yields of 1950 had been unchanged through 2000, a total of 1.8 billion ha of land would have been converted to cropland to meet the cereal demand. Instead, crop yield increases accounted for 77% growth in agricultural production between 1961 and 2005. Expansion into arable land accounted for only 14% of growth (Figure 3).

Increasing income

Increases in income and changing food preferences are impacting the quantity and type of foods that the world consumes. Over the past 60 years, the global average annual per capita growth of food and fiber consumption due to income growth was 0.27% (Buchanan *et al*., 2010). The growth in developing countries is much higher than is the case in high-income countries (Ibid). Increased incomes have particularly increased the demand for livestock products, fruits and vegetables. Delgado *et al*. (1999) estimated that the demand for milk in developing countries will increase annually by 3.3% from 1993 to 2020. Southern Asia will account for 60% of the increase while SSA will account for only 17% of the increase (Ibid). Increasing demand for livestock forces “extensification” since the demand for land and water for livestock is much bigger than for crops. For example, while 15 m^3 of water are required to produce one kilogram of beef, only 0.6 m^3 of water is required to produce a kilogram of cereals (Figure 7).

FIGURE 7 Water requirement of crops and livestock products.



Source: SIWI (2005).

Urbanization

Urban areas are growing fast and increasingly occupying larger land areas. The urban population surpassed the rural population in 2008 (UN 2008; Tollefson 2011) and it is expected that by 2050, the urban population will account for 70% of the total global population (Seto and Shepherd 2009). Cities occupy less than 3% of the global land area but they account for 78% of carbon emissions, 60% of potable water use and 76% of industrial wood consumption (Grimm *et al* 2008). However, measurement of urban area is not well captured by current LUCC models (Olson *et al* 2008).

Infrastructure development

Road development reduces transaction costs and increases access to natural resources. Hence, holding other factors constant, road development could lead to deforestation as observed by Nelson and Hellerstein (1997) in Central America. However, recent works have shown that such a pattern holds only in countries with weak institutions. In countries with strong institutions, road development does not affect deforestation. For example, between 1990- 2010, forest extent increased in Europe and China and has remained almost constant in North America (CBD 2010). All these countries and regions have good road infrastructure, strong institutions and high or middle incomes.

Studies have shown that access to road, electricity and communication infrastructure is strongly correlated with agricultural total factor productivity (Kamara 2008; Foster and Briceno-Garmendia 2010). This means that poor road infrastructure could lead to low agricultural productivity, which in turn could lead to the conversion of forest and other natural ecosystems to agriculture. However, poor road infrastructure could also hamper the cutting of forests for timber. For example, in Central Africa, the relatively low rate of deforestation is correlated with weak infrastructures. The SSA region has poor road infrastructure, which have contributed to low agricultural productivity, which in turn has led to conversion of virgin land to agriculture. As shown in Table 1, SSA has a large infrastructure deficit as compared to other developing countries. Such large deficit contributed to the smallest contribution of yield increase to agricultural production in 1961-2005 and to the fastest decline of per capita arable land (figure 3).

TABLE 1 Africa's infrastructure deficit and cost.

	Africa	Other developing countries
Paved road density (km/km^2 of arable land)[a]	0.34	1.34
Population with access to electricity (%)[a]	14	41
Population with access to improved potable water (%)[a]	61	72
Power tariffs ($/kwh)	0.02-0.46	0.05-0.1
Transportation cost ($/ton/km)	0.04-0.14	0.01-0.04
Tariffs of urban potable water ($/cu m)	0.86-6.56	0.03-0.6

Note: [a]Excludes medium income African countries (South Africa, Kenya, Botswana, Gabon, Namibia, Cape Verde, etc.) and is compared to other low income countries. The rest of the statistics refers to entire Africa and other developing countries.

Source: Foster and Briceno-Garmendia (2010).

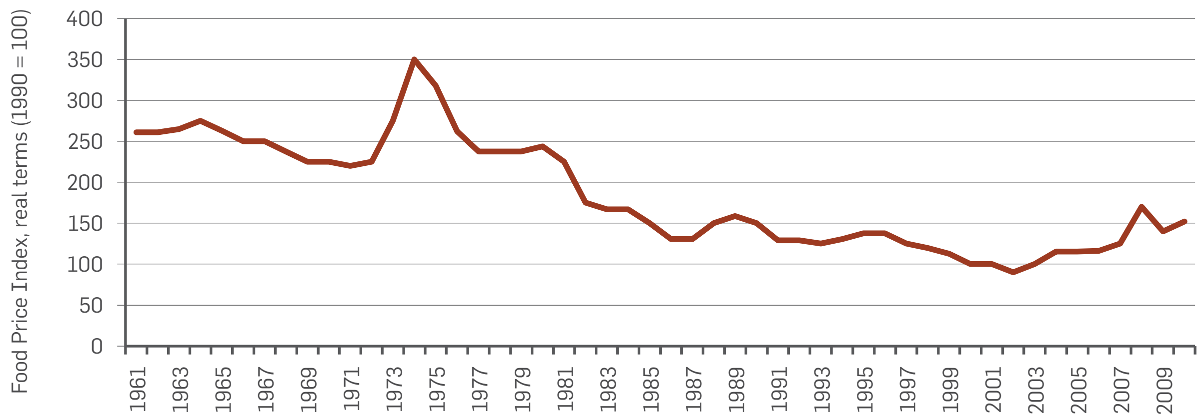
Access to markets and information helps land users to make informed decisions. Farmers with access to market information will respond to market signals and could respond favorably when they have better access to market and information. For example, the famous "more people less erosion" study in Kenya by Tiffen *et al* (1994) was a result of better market access by land users in Machakos district, which is only 54 km away from the city of Nairobi (Boyd and Slaymaker 2000).

Food prices and price elasticity of demand

Food prices provide incentives for farmers to convert land to food production. The recent high food prices and the consequent land grabbing illustrate this pattern. Large international deals were made following the food price spikes in 2007-08 and 2010-11 (Figure 8), which were driven by a variety of factors that vary across the different agricultural commodities. Among these are investment-driven speculation, the government policy response and the news media (Andersen and Watson 2011) as well as the rapid increase in biofuel production from maize in the U.S. (Rosegrant *et al* 2008; Baffes and Hanniotis 2010), particularly in the run up to the 2007-08 price spike. A more detailed discussion of the factors underlying the 2007-08 food price spike is given by Headey and Fan (2011), who synthesize a number of findings that have emerged in recent years as more data has become available. The trend that has been observed in which rich countries with arable and/or water deficits acquire lands in developing countries with abundant arable land and/or water has caused concern among policy analysts and researchers (von Braun and Meinzen-Dick 2009; von Braun 2011) as to the socio-economic and institutional implications for small-holders (Bomuhangi *et al*. 2011) and women (Behrman *et al* 2011). Additionally, several investment funds have indicated their intention to invest more than US$ 2 billion in land for food production in Africa (Ibid). Globally, about 46.5 million ha were acquired between 2004-2009 in 81 countries (Deininger *et al* 2001; Toulmin *et al* 2011). Such large land deals change land use. Many recent land deals, for example, have displaced community-managed lands which combine shifting cultivation, livestock, and forest resources (Toulmin *et al* 2011) with monocrop systems or other large-scale production systems -- which in turn reduce biodiversity. A recent study observed that foreign land acquisition was more likely to occur in countries with abundant land and weak land governance, supporting the growing concern regarding the lack of protection of vulnerable groups against foreign land acquisition (World Bank 2011a).

Price elasticity of demand for food also drives LUCC. Food demand is price inelastic in high-income and fairly elastic in low-income countries (Hertel 2011).This means high food prices will lead to greater incentives for producers to convert land.

FIGURE 8 Global Food price Index trend, 1961-2010.



Source: FAOSTAT.

Policies at national and international level

Policies both at national and global level have a large influence on LUCC. Recent studies have shown that increasing food prices have prompted importing countries to change their trade policies to protect consumers while exporting countries have changed trade policies to the benefit of farmers. For example, the Global Trade Alert (http://globaltradealert.org/) found that 45 food exporting measures and 85 import measures were changed between November 2008 and November 2011. The impact of such trade policy changes on the international rice, wheat and maize international prices were estimated to be respectively 31%, 13% and 18% (Table 2).

TABLE 2 Contribution of domestic food policies (trade tax) on international price spikes of major crops, 2005-08.

Crop	Total (Percent)	Of which:		Of which:	
		High-income countries	Developing countries	Importing countries	Exporting countries
Rice	31	1	30	13	18
Wheat	13	6	7	6	7
Maize	18	8	10	7	11

Source: Anderson and Nelgen (2012).

The impact of the price change due to such policies could be felt through the price impact on LUCC and through the direct impact. Minimizing the negative impacts of country-level policies on global or regional community requires a global action through the World Trade Organization (WTO) and other forms of international cooperation (Andersen and Nelgen 2012 and Martin and Anderson 2012).

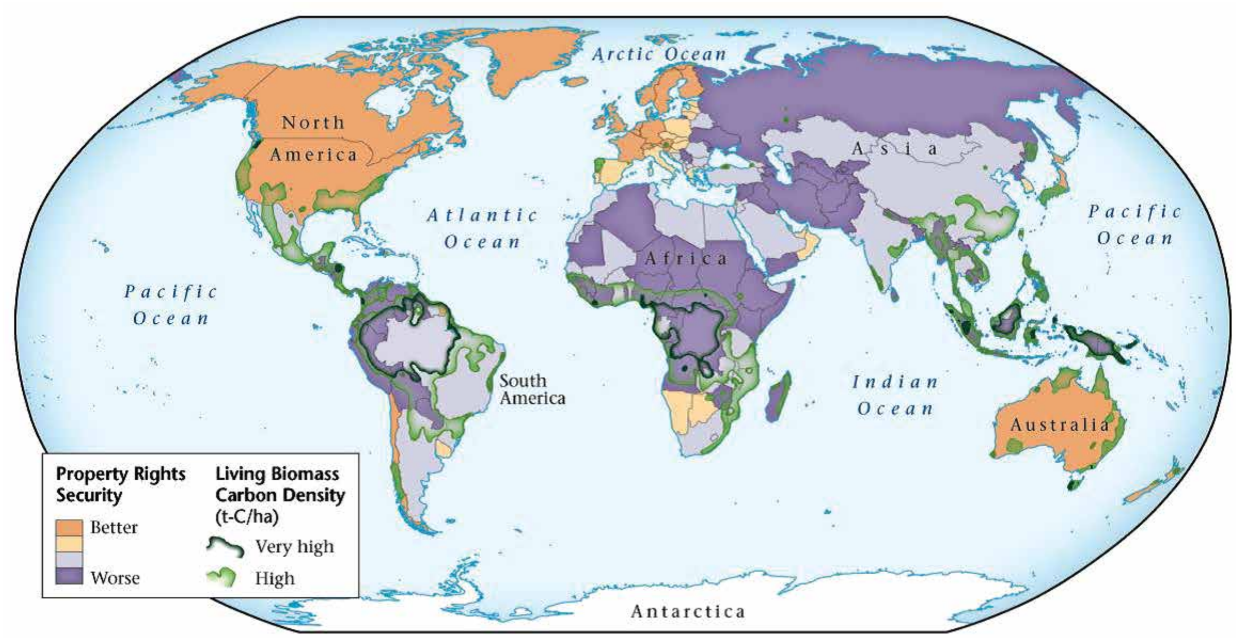
Countries have also used policies that encourage farmers to use or not to use land or to improve or degrade land. For example, the U.S. Conservation Reserve Program (CRP) fallowed 12.5 million ha in 2005 (Wunder *et al* 2009; Claassen *et al* 2009). As will be seen below, systems of payment for ecosystem services implemented by countries and the international community have shown promising results of land improvement and LUCC in general. However, the benefits of some PES programs have been questioned. For example, Wünscher *et al* (2008) observed limited additionality of PES in Costa Rica due to the country's low deforestation rate.

Land tenure and property rights

Tenure security provides incentives for long-term land investments such as tree planting and soil and water conservation structures (Feder 1987; Alston *et al* 1995) and LUCC. However, it has also been established that land holders with insecure tenure may plant trees or engage in other long-term investments to enhance their security or as a method of claiming ownership (Place and Otsuka 2002; Braselle et al 2002). Other studies have also shown that investments by farmers with customary land tenure were comparable or greater than investment by farmers holding land with secure title deeds (leasehold or freehold) (Toulmin and Quan 2000; Deininger 2003). Additionally, secure land tenure is a necessary but insufficient condition to determine investment or LUCC. Other factors driving LUCC and land investment incentives (e.g. those discussed above) play a key role.

Global studies have shown diverse systems of land ownership, tenure, and land rights exist across continents, with different degrees of tenure security. A recent study by Bruce *et al* (2010) showed that areas with strong land tenure security have relatively lower living biomass carbon density than areas with rich biomass carbon density (Figure 9).

FIGURE 9 Relationship between land tenure security and living biomass carbon density.



Source: Bruce *et al*. (2010).

Bioenergy

Bioenergy places a new pressure on land demand. The global growth in biofuel comes in the face of increasing scarcity of energy resources and growing energy demand for transport fuel and other productive uses. A number of OECD countries have engaged in large-scale biofuels production as a way of exploiting renewable resources to supplement and diversify their domestic energy portfolio. North America has been the largest consumer of biofuels, worldwide, followed by Latin America and the European Union (IEA, 2008; von Braun 2008). Together, Brazil and the U.S. account for over 90% of the world's ethanol production; the U.S. overtook Brazil as the world's leading producer of ethanol in 2004. Biodiesel, on the other hand, is mostly concentrated in the EU (IEA, 2008). Besides the desire for enhanced energy security and diversification, a major policy motivation for biofuels production has also been to reduce greenhouse gas (GHG) emissions from fossil fuels, especially in the EU. The actual GHG emissions savings, however, depends heavily on the production pathway, and is a source of active debate and research.

The extent of land use changes that are caused by large-scale biofuels production has generated a great deal of debate within the energy and environmental policy and research communities. A recent study showed that of the 203.4 million ha of land acquired globally since 2000, 66% was obtained from Africa and that of the 71 million ha verified by the study, 40% were acquired for biofuel production while only 25% was for production of crops for food, 3% for livestock production and 5% for non-food crops such as cotton (Anseeuw *et al* 2012). This reflects the land competition and potential for compromising food security efforts in Africa – the world's most food insecure region—and carbon sequestration if such acquisitions are located on forest land. Hertel (2008) estimated that U.S. and EU biofuel mandates will increase crop land cover at the expense of forest and pasture cover (Table 3).

TABLE 3 Predicted change in global land use due to US & EU mandates.

	US	EU-27	Brazil
	2001-2006 (% change)		
Crop	0.3	0.7	1.1
Forest	-0.7	-2.1	-2.6
Pasture	-1.4	-2.3	-2.2
	2006-2015 (% change)		
Crop	0.8	1.9	2
Forest	-3.1	-8.3	-5.1
Pasture	-4.9	-9.7	-6.3

Source: Hertel *et al* (2008).

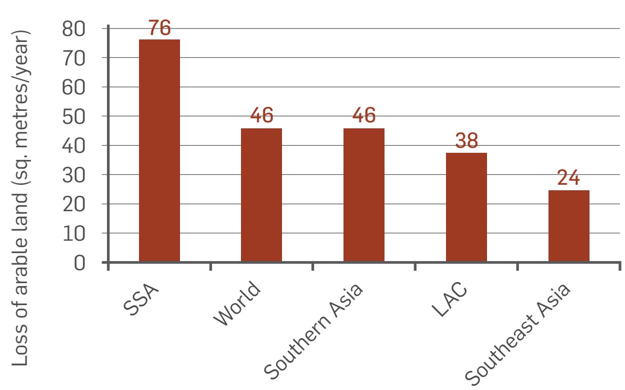
Additionally, the mandates will lead to greater use of fertilizer and other agricultural inputs, which in turn could lead to environmental pollution. For example, Britz and Hertel *et al* (2011) estimate that by year 2015, nitrogen fertilizer use will increase by 0.14% in EU-27 due to EU bionergy mandates compared to its level in 2001.

Land degradation

Land degradation, defined as loss of the capacity of land to provide ecosystem services, affected about 24% of the global land area between 1981 and 2003 (Bai *et al.*, 2008). This is equivalent to a degradation of about 1% of global land area each year or about 12 Mha (UNCCD, 2011). This area could produce 20 million tons of grain each year or 1% of the global annual grain production of 2.241 billion tons (UNCCD 2011; USDA 2011). Globally, 1.5 billion people live on degraded lands. It is also estimated that 42% of the very poor live on degraded lands (UNCCD, 2011).

Land degradation reduces both land productivity and arable land area. Land area is reduced when land is degraded beyond productive level. A reduction in the productive capacity of land leads to agricultural expansion into forests and other natural ecosystems. Land degradation could also change land use. For example, it is common for farmers to turn highly degraded cropland into grazing land. Increase in population density also contributes to land degradation in developing countries when farmers continuously cultivate land without adequate replenishment of soil nutrients. The per capita arable land area in SSA has decreased more than in any other regions in the world (Figure 10). Of particular importance is fire, which has a large impact on land cover. Human-induced and natural fires all change land cover significantly. The ability to monitor fires using high frequency satellite observations has improved over the years (Giglio *et al* 2009). In 2000, human-induced fires accounted for 3.2% of HANPP (Harbel et al 2007). Naturally occurring fires also alter land cover; it is estimated that there are 200 million ha of lands, mainly in the far northern boreal forests, that have been degraded by wild fires (Minnemeyer, et al 2011). All this underline the importance of land degradation in LUCC and how its prevention could help address the overall impact of land use change.

FIGURE 10 Trend of loss of arable land area per capita across regions, 1961-2009.



Key: SSA = Sub-Saharan Africa; LAC = Latin American and Caribbean countries.

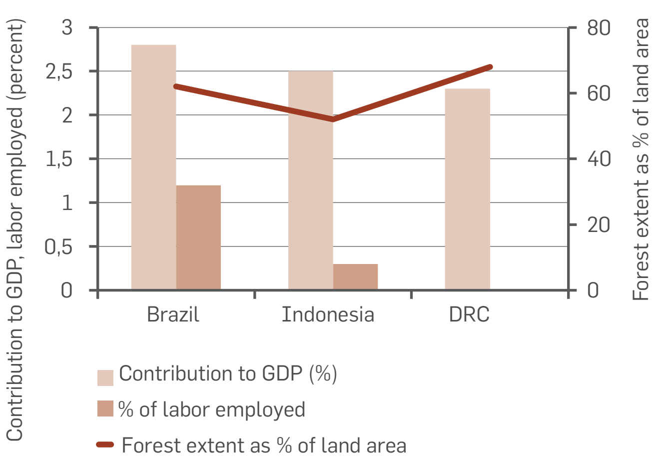
Source: Calculated from FAOSTAT data using trend line regression.

2. *How land use change happens in practice, and how competing demands on land are managed*

Introduction

Since LUCC patterns and trends and their drivers vary significantly across countries, we use three case study countries to better understand their dynamics and interrelationships. We use Brazil, Democratic Republic of Congo (DRC) and Indonesia, three countries with large rainforests that account for more than 50% of the land area (Figure 11) and are at different stages of development. Brazil accounts for 13% of the 2010 global forest extent of 4,033.06 million ha while DRC and Indonesia respectively account for 4% and 2% of the global forest extent (FAO 2011a). The Brazilian agricultural sector has been a unique example; its contribution to the GDP has increased from 5% in 2006 to 6.1% in 2010 (World Bank 2011c) while its deforestation rate has fallen dramatically. DRC is home to the largest rainforest in Africa. With 68% of its land area under forest (FAO 2010), the country accounts for 34.6% of the region's carbon stock (Baccini *et al* 2008). However, forest contributed only 2.3% of DRC's GDP in 2006 (FAO 2010). Forest and agriculture in Indonesia drive land use change. Forest is the largest land use type in Indonesia. The extent of forest in Indonesia covers about 53% of land area and Indonesia has the third largest tropical forest (FAO 2010)[3]. The sector contributes about 2.5% of Indonesia's GDP (Ibid). The agricultural sector, which contributes 16% of the GDP (World Bank 2011c) covers only 22% of the land area (FAO 2010).

FIGURE 11 Contribution of the forest sector to GDP and employment in case study countries.



Note: Percent of labor employed in forest sector in DRC is not available.
Source: FAO (2011a).

Land management in Brazil

Main interests at play

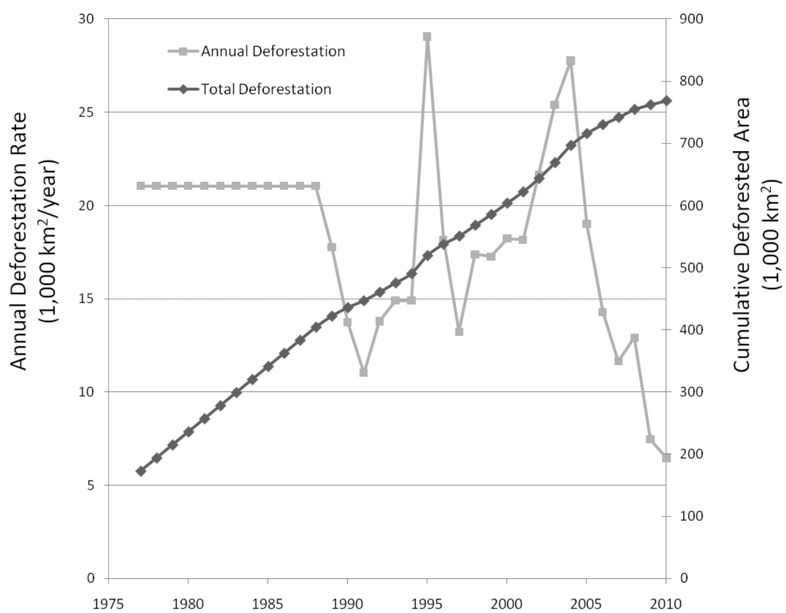
Occupation of the Brazilian Amazon, hereafter called the Amazon, by humans dates several thousand years. For most of the time, human occupation did not affect the integrity of the Amazon ecosystems, but in the last four decades anthropogenic change has proceeded at an unprecedented pace, putting at risk hydrological and biogeochemical natural cycles. Land occupation of the Amazon has been driven by several players or agents of changes operating at different intensities and in different areas and time periods. There are also multiple interests by these agents of change that can be summarized in four groups.

The first group is formed by federal and state governments, large construction firms and politicians who have interest in accelerated development in the region. Several governmental development programs were established in the past four decades to construct and pave roads (Fearnside, 2002; Peres, 2001), build hydroelectric dams to supply the energy demand in the country and invest in the development of industrial and mining activities. The second group is crop farmers and ranchers who were supported by the government in the form of subsidies and bankers, traders and politicians who provided credit and other financial services to these activities. The third group includes settlers in public unoccupied territories who harvest timber and small-scale miners. The fourth group is the leaders of local environmental sustainable development programs, advocacy groups and the international community, all of whom aim to promote the environmental conservation and maintenance of ecosystem services and cultural diversity (Foley, 2007). The national and local-level sustainable development and advocacy groups include NGOs, governmental agencies, indigenous and traditional communities and progressive private companies.

The allocation of timber concessions to the national forest (Flonas) is also a major objective of the government for the sustainable management of forests and to avoid de facto open access. Up to 35 M ha are targeted for timber concessions.

Due to the interaction of these groups, the Amazon ecosystem has undergone rapid changes in the past four decades. Until 2011, 762,000 km^2 of natural forests were converted to predominantly cattle ranching and crop production (Inpe, 2009). This implies that 19% of the original Amazon forests were cleared. The pace of forest conversion varies through years, but in the last three decades (i.e. 1980-2009) official statistics of deforestation shows that 4-5% of the original forest was lost per decade. Most of the forest conversion (72%) happened from 1980 to 2011 (Figure 12). Additionally, selective logging, which is mostly predatory and illegal, and forest fires affect an area of the same magnitude of deforestation (Asner, 2005; Peres, 2006).

FIGURE 12 Annual deforestation rates and cumulative deforested areas in the Brazilian Amazon Biomes.



Source: Inpe, Prodes Project @ http://www.obt.inpe.br/prodes/.

Favorable conditions to initiate deforestation at a large scale in the Amazon were set in the late 1960s. By that time, the region had very low human population density and this made the government to promote its occupation with new road infrastructure and settlement in uninhabited territories. This was done in order to develop and integrate the region to the other parts of the country. The government also aimed at consolidating Brazilian sovereignty over its vast territory of 5 million km^2. The National Integration and Land Redistribution plans were the main policies implemented to achieve these goals. As a result of this, fast development and occupation process, deforestation rapidly increased in the 1970s, mainly within human settlement areas and along the main axial roads (Cuiaba-Porto Velho, Santarém-Cuiaba, Belém-Brasília and Transamazonica Roads).

In the 1980s, the government continued to play a major role in pushing the development frontier of the Amazon region by providing subsidies to cattle ranching and small-scale agriculture, and maintaining infrastructure investments. Long-term tax reductions provided incentives to developers of large- scale mines. Hydroelectric dam construction also played an important role in attracting more people to low density occupation areas. So, too, did the spontaneous occupation of large territories by gold miners in areas like the Tapajós basin and Marabá Serra Pelada in Pará. Logging activities, especially for very high-value tree species like mahogany, was also scaling up, damaging vast areas of forests due to illegal tree harvesting and construction of large road networks (Verissimo, 1995). It was when these activities were underway that the first alarming satellite images showed the dangerous speed of rainforest destruction. By the end of 1980s, 420,000 km^2 had been deforested (Figure 12).

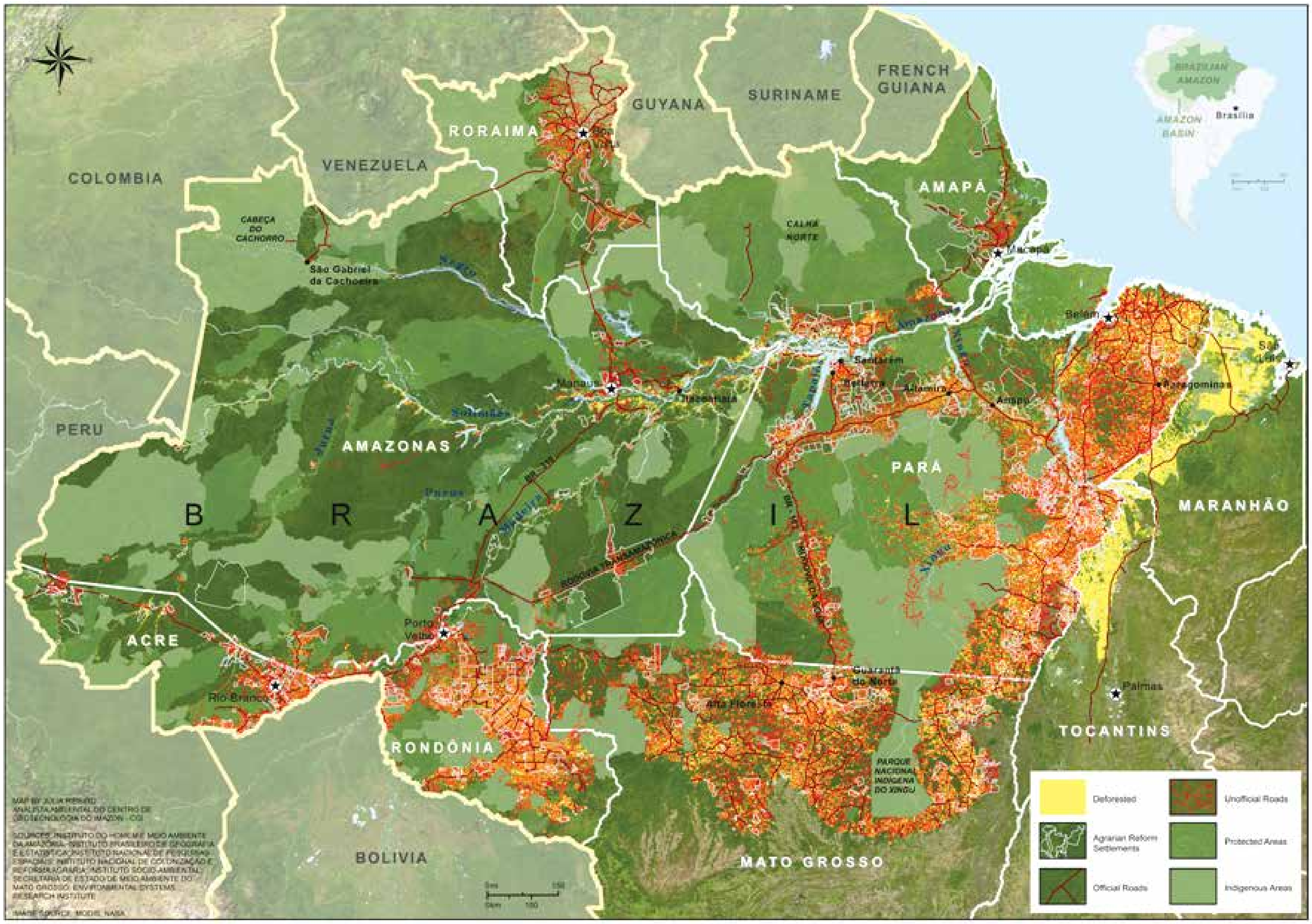
In the beginning of the 1990s, an NGO movement emerged in Brazil as a result of growing socio-environmental concerns. By this time, there was a decline in government investments that encouraged the occupation of the Amazon; nonetheless, the region's abundance of natural resources, such as timber and gold, and land availability, accelerated the region's unofficial occupation (Uhl, 1997). Logging roads opened in the late 1980s, providing access to new settlements and leading to the appropriation of large areas for mainly cattle ranching. The logging boom brought ephemeral economic prosperity to dozens of milling centers in states throughout the Amazon. Milling centers like Paragominas, in eastern Pará had 240 sawmills that produced more than 100 thousand cubic meters of processed timber, generated thousands of jobs, and generated high taxes (Verissimo, 1992). However, due to a decline in timber resources, sawmills had to close and move to other areas where timber was still abundant. This classic illustration of a boom-and-bust cycle impacts unsustainable logging activity—as well as any activity over-exploit its resources (Rodrigues, 2009).

Combining data from deforestation, logged areas, rural settlements, major cities, and official and unofficial roads, provides a clear picture of the extent to which the Amazon is occupied by humans (Figure 13). The occupation frontier goes beyond deforestation boundaries due to the extensive network of (illegal) roads mostly for logging, mining operations, and settlement that can be detected by satellite images (Brandão, 2006). Activities like hunting, exotic species invasion, non-timber forest products harvesting (NTFP), and others, are almost undetectable by satellites, making the area impacted by humans much larger. Some 44% of the Amazon is protected (Verissimo, 2011). Throughout Brazil, there has been an increase in protected areas since 1992 (Figure 14). These important areas work as an effective buffer against deforestation expansion, but illegal logging and mining are still active if declining due to greater efforts by the government to enforce the protected area and provide reward mechanisms for conservation, which is discussed below.

Government control on LUCC

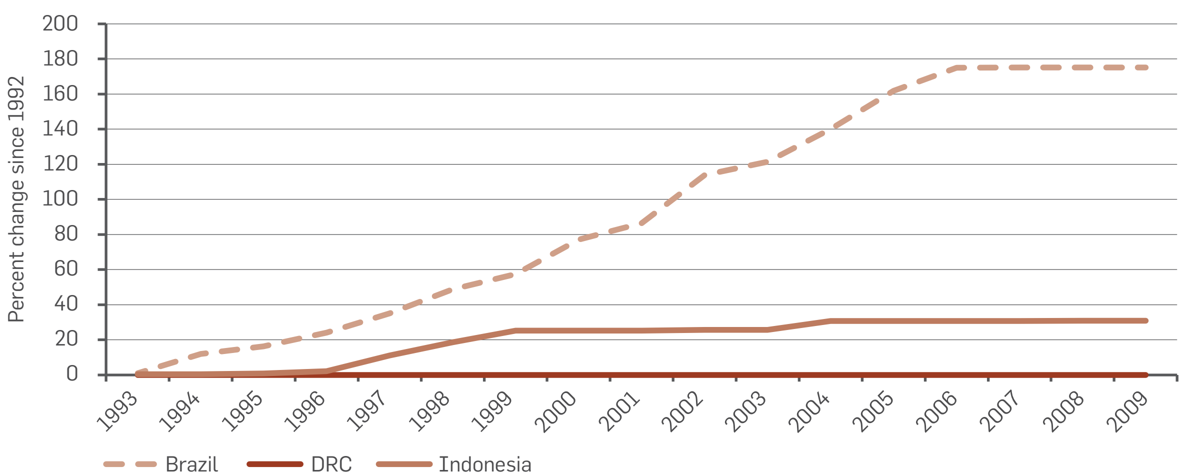
Federal, state, and municipal governments have taken actions to stop deforestation through the implementation of important measures. First, the Protected Areas as Conservation Units was expanded. With support from Brazilian and International NGOs, the Ministry of Environment created 487,000 km^2 of new Protected Areas between 2003 and 2006 (Verissimo, 2011). Second, the government imposed stronger enforcement of the rules prohibiting deforestation. Third, the government created a list of municipalities that most contributed to deforestation and imposed restrictions on access to credit for agriculture activities in those areas. Payments for ecosystem services were also provided to incentivize land users to stop deforestation. If these land users protected trees and adhered to the deforestation moratorium conditions, they were paid from funds generated through government programs and international carbon offset funds (Nepstad *et al.* 2010). Major International donors involved in the Amazon's protection include the Worldwide Fund for Nature (WWF) and the Germany and Global Environment Facility (GEF) (CBD 2010), among others. As Figure 12 shows, deforestation rates started to decline dramatically from its highest annual rate of 72,000 km^2 in 2003-2004 to only over 7,000 km^2 in 2008-09, a 74% decrease in only five years (CBD 2010).

FIGURE 13 Modern human occupation in the Brazilian Amazon and Protected Areas in the Region.



Source: Imazon.

FIGURE 14 Trend of protected area in Brazil, DRC and Indonesia since 1992.



Note: Change of protected area in DRC increased less than 0.01% throughout the time under consideration.
Source: IUCN and UNEP-WCMC (2011).

Pattern of LUCC in the Amazon and government objectives

The lessons from the last three decades show that forest cover change and land use in the Amazon region were mainly driven by large-scale governmental investment policies. Federal efforts to develop and occupy the Amazon region from the 1960s to the early 1980s through road construction, subsidies to cattle ranchers, and tax breaks to miners and hydroelectric power developers created incentives for the occupation of the Amazon. The government also played and continue to play an important role to control deforestation by restricting access to unoccupied lands by creating Protected Areas, conducting intense and permanent enforcement of its new policy aimed to prevent deforestation and promote sustainable land use practices. This has shown significant progress and deforestation reached its lowest level since 2004 (Figure 12). Brazil has also put forward a pact to reduce deforestation by 80% by 2020. The international community, NGOs, news media and independent opinion leaders have also been key to raising awareness to the general society.

Market forces have also played an important role. The payment of soybean producers not to clear forest (the so-called soy moratorium) demonstrated the importance and effectiveness of market forces in controlling deforestation. However, an evaluation of this program is needed to identify strategies for preventing leakage and other weaknesses.

In summary, the dramatic story of the Amazon's increasing then decreasing deforestation demonstrates the role played by government institutions and the effectiveness of the incentive mechanisms implemented in the last five years. However, much remains to be improved. In a recent study Fearnside (2011) found that taking the exchange rate and relative prices into account matters. He found that recently, the international price of beef and soy fell, cutting the profits of commodity exporters deeply (Fearnside, 2011).

Indonesia

Interests at play

Agriculture and forest are the two major land types with stiff competition in Indonesia. Between 1990-2000, the country lost 1.9 million ha per year or 1.7% of its forest cover (FAO 2011a). According to Taconi (2003), however, forest fires destroyed 11.7 million ha during the 1997-1998 el-Niño year-- the most significant el-Niño damage in the world. The palm industry has been one of the driving forces of such loss and Indonesia is among the few countries with a large extent of tropical forest with high deforestation rates (Grieg-Gran 2008). Commercial logging is also a major problem in Indonesia that dates back to the 1970s. Between the 1970s and 2000, some 60% of Indonesia's 100 million ha of forest was allocated to commercial logging, which led to an annual log harvest of about 70 million m^3 (Barr 2001) — well above government sustainable harvesting level set at 25 million m^3 (Casson 2001). One reason behind this unsustainable harvesting was the 1998 decentralization of the forest sector (Casson 2001), which involved attempts by local governments to increase revenue and legalize logging (Barr 2001). In 2000, for example, the KotawaringinTimur district collected US$ 6.2 million from natural resources, more than half of which came from illegal logging (Casson 2001). However, decentralization only compounded the deforestation that had existed long before 1998 due to logging industry corruption, including the subcontracting of timber exploitation with perverse incentives (progressive payments for timber volume supplied by the contractors to the concessionaire) and overcapacities in the wood processing industry resulting in massive amounts of illegal logging.

Despite such an active lumber industry, agriculture remained the leading cause of deforestation in Indonesia (Dechert *et al* 2005). Bearing in mind the impact of deforestation and in response to REDD+ initiatives, the government entered into a contract with the Norwegian government to suspend all concessions for two years for the conversion of peatland (partially decayed organic matter in wetlands) and forest areas to other uses (Murdiyarso *et al* 2011). This was part of preparations for the REDD+ National Strategy, which was announced in 2009.Under this strategy, the Norwegian government committed to providing US$ 1 billion to protect about 7.2 million ha (Mha) of primary forests, 11.2 Mha of peatlands, and 4.1 Mha of other types of conversions (Ibid). This was in line with the government's voluntary target of reducing GHG emissions by 25% by 2020 (Ibid).

The current moratorium does not cover secondary forest or logged over forest and it excludes conversions for food and energy security, thus creating loopholes (Ibid). Moreover, enforcement of this moratorium is still uncertain given general weak enforcement and governance. As in the case of Brazil, the Indonesian forest sector program shows the key role played by commitments from both the national government and the international community. The annual deforestation rate fell from 1.7% in 1990-2000, one of the highest in the world, to only 0.5% in 2000-2010 (FAO 2011a). One reason for the reduced deforestation is an increase in the protected area. Compared to its level in 1992, the protected area increased dramatically in 1997 and in 2004 (Figure 14), a trend that reflects the land reforms (reformasi) implemented by the regime that replaced President Suharto in 1998.

The Indonesian government has also implemented other policies to manage forest and land, including the Kamasyarakatan (HKm), which provides farmer groups with permits to continue farming on deforested state land designated as Protection (or Production) Forest (PF) in exchange for sustainable forest management. This approach contrasts with the past approach, which included the forcible eviction of farmers who encroached on government-owned forest lands (Pender *et al* 2008). The implementation of the HKm program is innovative as it empowers local communities to manage state-owned forest land. Communities are required to comply with forest management laws and use participatory decision making and conflict resolution. The management permit also specifies areas for crops and forests. For example, the HKm mandates that communities should set protection blocks of natural forests within 500m of a dam or lake, 200m from a water spring, or 100m from a riverbank or land with a slope of more than 40% (Pender *et al* 2008). On cultivated areas, farmers are required to use practices that won't lead to soil erosion and other forms of land degradation. Communities holding HKm are also responsible for protecting the forest area from fires, illegal encroachment, and other threats. An evaluation of the HKm program by Pender *et al* (2008) revealed that it resulted in the increased planting of timber and multipurpose trees.

Forest fires in Indonesia also pose a major challenge to the conservation of tropical forests. These fires are often the result of attempts to establish plantations: it is easier to obtain a declassification of the permanent forest estate when part of it has been burned. According to potential future climate scenarios, conditions could arise that could increase wildfire hazards in tropical rainforests. To control wildfires, the Indonesian government has enacted a number of laws and regulations. Compliance with these regulations, however, has generally been poor due to the low capacity of national and local institutions to enforce them (Herawati and Santoso, 2011).

Overall, Indonesia's HKm and other land use policies demonstrate the long-held view that the participatory involvement of local communities in the management of forests and other natural resources promises cost-effective achievement. Such approaches have worked in other countries and have contributed to forest recovery not explained by economic growth in the forest transition model (Foster and Rosenzweig, 2003; Fang *et al* 2001; Ruddel *et al* 2005).

Democratic Republic of Congo

DRC, the largest Congo Basin country, harbors 114 million hectares of dense forests, with an additional 23.7 million of driest woodland forest called *Miombo* and almost 37 million hectares of wooden savannah (Figure 15; de Wasseige *et al*, 2012). DRC, SSA's third most-populated country[4], has a population of 70 million and a demographic growth rate of almost 3% per annum. As DRC has not yet entered the demographic transition stage, the population is expected to reach 120-130 million in 2030 at the current rate. With a GDP per capita estimated at around $130 in 2006, the DRC is one of the world's poorest countries despite huge reserves of natural resources, especially mining products. In 2010, about 71% of the population lived below the national poverty line (UNDP, 2011); the 2011 Global Hunger Index placed DRC at the lowest level in the world (Grebmer *et al*, 2011). A global ranking of government effectiveness (Kaufmann, Kraay and Mastruzzi, 2007), a government's capacity to implement policies with independence from political pressures and with respect to the rule of law, put the DRC's government as the fourth least-effective (World Bank, 2009).

A brilliant agricultural past and a small industrial timber player

In 1912, the Lever Brothers of Unilever established its first oil palm plantation in the Congo. After the Second World War, the country was among the largest exporters of perennial crops in the world. In 1958, Congo became the world's first palm oil exporter, a position eventually taken over by Nigeria (Tollens, 2004). Total industrial production of the oil is estimated today at 25,000 tons. Presently, the DRC imports palm oil from Asia—about 80,000 -100,000 tons per year and growing. "Zaïrianisation" (nationalization) and the accompanying civil war caused dramatic agriculture decline, with thousands of hectares of plantations abandoned or looted. The Congo still has the natural potential to become again an agro-industrial powerhouse, but the institutional context means that this goal is quite unlikely to be achieved in the short term.

Industrial forestry — despite its potential in the country — has always been underdeveloped in Congo. Although most Congolese forests are not as rich, in commercial terms, as those in neighboring countries, this underdevelopment is due to a lack of infrastructure. The country's road network is one of the poorest in Africa: The country received the lowest "road transport quality index" rating in the region. In contrast to Cameroon (18.4) and South Africa (100), it received an 8. Worsening the situation is the fact that the Congo river cannot facilitate the transport of logs to the Atlantic port of Matadi from the upstream city of Kinshasa; instead, the timber has to be trans-shipped from river boats to trains or trucks. The port itself, built in an estuary, cannot host large vessels and its warehouses are always saturated. As a result, transport costs are very high. Carrying a cubic meter of timber from Kinsangani to Matadi costs $120-150, takes four weeks, and involves the significant risk of losing cargo (Debroux *et al,* 2007). Hence, loggers who target exports only focus only on a handful of high-value species and, on average, they harvest only 4-5 m^3 of timber per hectare (commercial volume), one of the world's lowest rates. The registered timber production is around 300-350,000 m^3 per year, and, in the past, had rarely exceeded half a million. However, there is a large informal production run by chainsaw loggers and small-scale enterprises for domestic markets and illegal export to countries east of DRC (Uganda and Rwanda, for example). Imprecise estimates put the production at 4-5 million m^3 a year for this informal (and thus illegal) activity (Djire, 2004). An unknown volume of timber is trafficked across DRC's eastern borders by informal but relatively large enterprises that are not part of the traditional timber industry sector. This traffic could represent several hundreds of thousands cubic meters per year, possibly more than the exports registered from the Atlantic side. It seems that this phenomenon has picked up since the 2009 construction, by the China-DRC cooperation, of the Kisangani-Béni 750 km-long road (from the north-center to the far east of the country, close to Uganda's border).

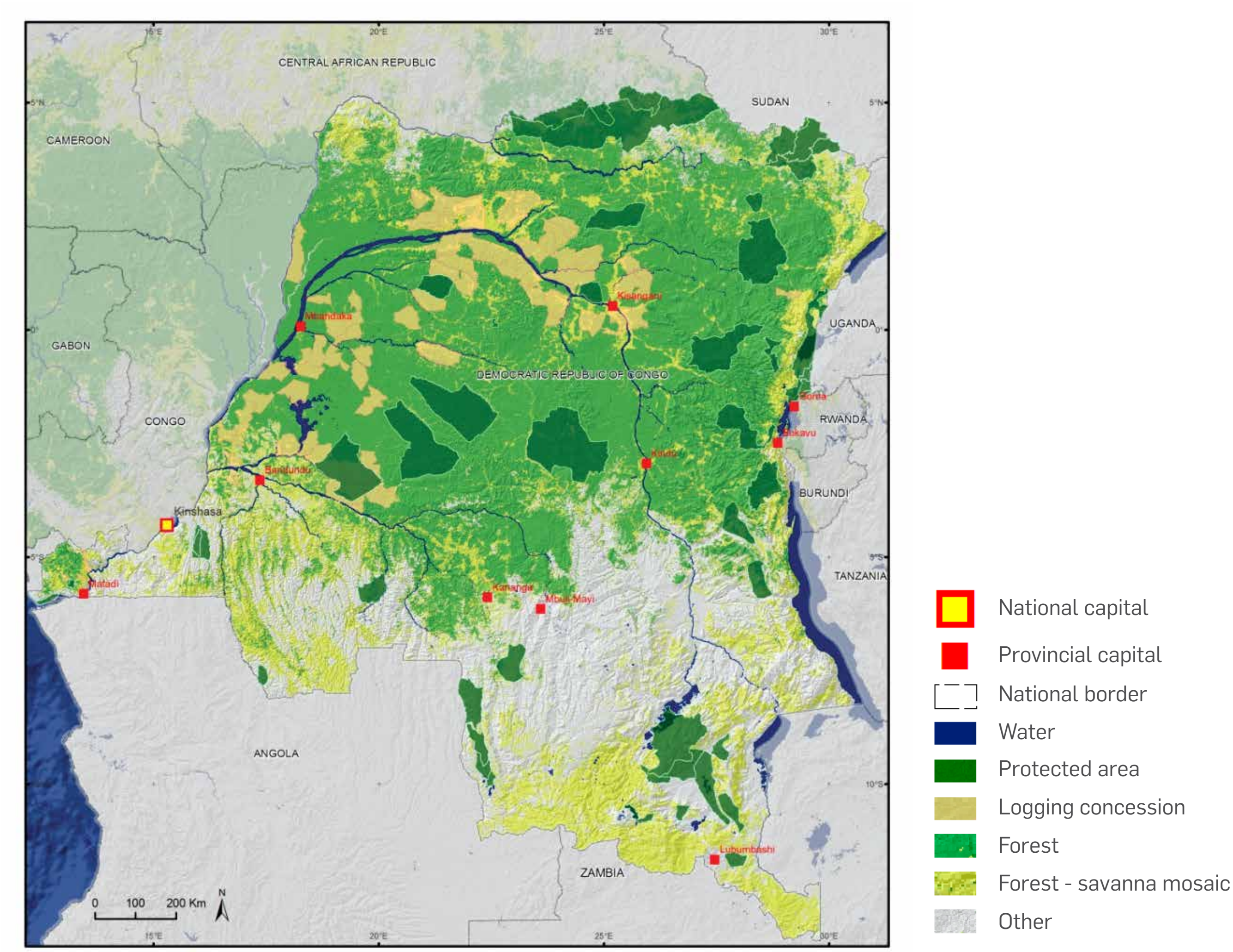
The area covered by legal concessions has dramatically declined since 2000, when 42 million hectares were allocated (Figure 16). The introduction of an area tax in 2002 led many permit holders to abandon vast areas of concessions they kept unexploited for speculative reasons. After a period of illegal allocations in 2003-2005, the DRC decided to engage in a review of the validity of the forest titles. At the end of this process, in 2009, the surface covered by legal concessions dropped to 12 million hectares (Mertens and Bélanger, 2010). There is still a moratorium—set to last through 2012 at least—on the allocation of new concessions. Due to the numerous difficulty faced by the enterprises in this "fragile state", the timber concessions, often operating in competition with small-scale loggers and farmers, are declining. In early 2012, one of the largest concessions held by German interests (1.3 million ha) closed down for not being profitable enough. The abandonment of large concessions does not mean the forest will remain untouched: chainsaw loggers and informal enterprises, which operate illegally even in running concessions, are expected to take advantage of the existing trails networks to increase their exploitation efforts.

Land use dynamics

Deforestation in DRC has increased with the progressive end (except in the East) of the country's civil war in 2000. The annual gross rate of deforestation for 1990-2000 was 0.15 % (0.11% net). It climbed to 0.32% (0.22% net) for the 2000-2005 period (de Wasseige *et al*, 2012). The most recent data indicate another increase for the 2005-2010 period, with almost 2 million hectares lost overall—up 13.8 % from the 2000-2005 period (Ernst *et al*, 2010).

In DRC, the main drivers of deforestation are informal logging for timber for local or regional use, charcoal production and land clearing by shifting cultivators (Tollens, 2010). In contrast, logging for legal export, plantation establishment and cattle ranching only contribute a small share of deforestation in the country (Ibid). Spatially-modeled analysis undertaken by Delhage *et al* (2010) found that one of the main drivers of deforestation is traditional smallholders' agriculture activities, which are dominated by roots crops (cassava, yams, and cocoyam) and banana and plantains. These have low productivity, are based on shifting cultivation and use minimal to no external. Population increase in rural areas, forest fragmentation and roads are compounding factors.

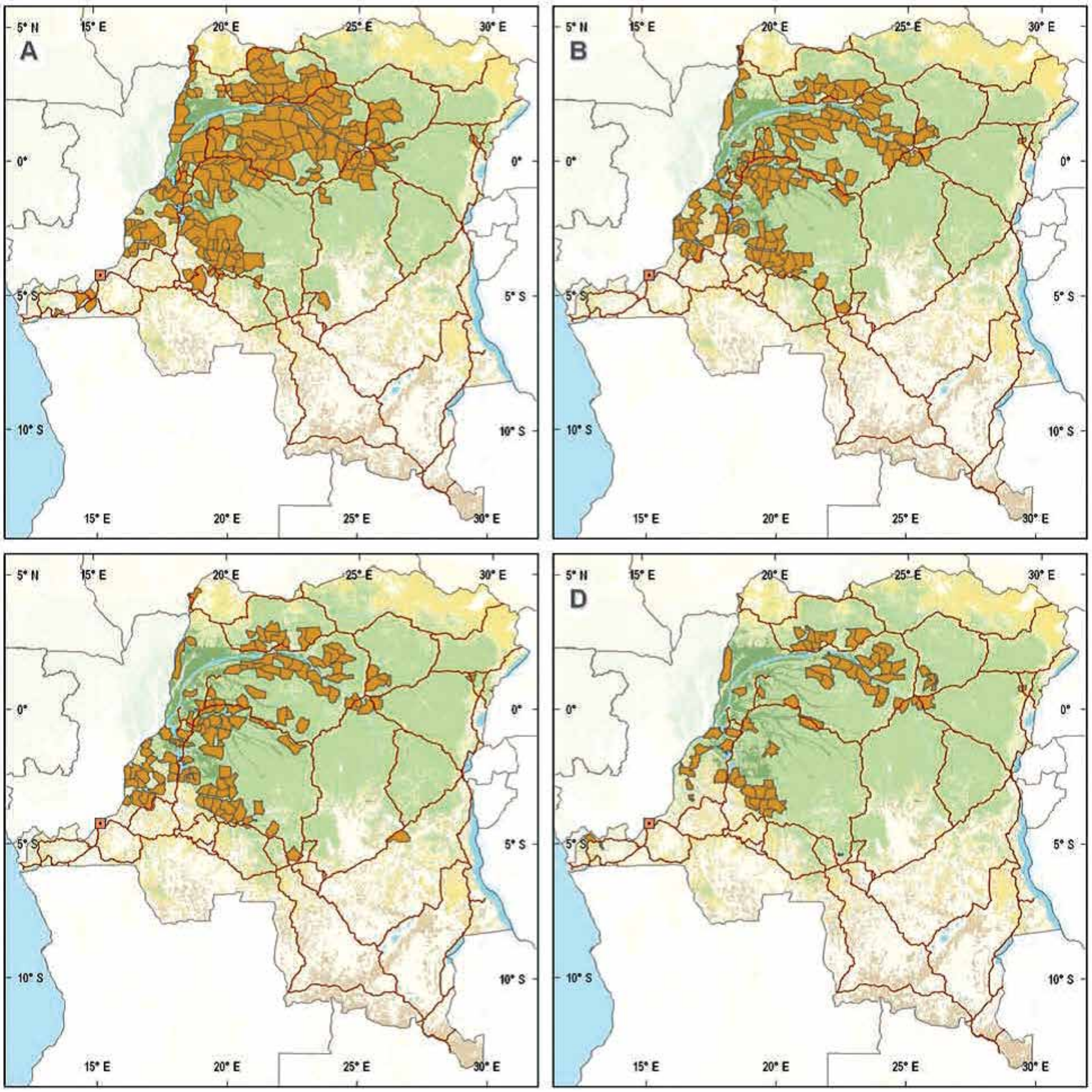
FIGURE 15 Concessions and protected areas in DRC.



Source: de Wasseige *et al* (2012).

FIGURE 16 Changes in the surface allocated to timber concessions.

	A	B	C	D
Year	2000	2003	2007	2009
Area (million ha)	42	25	26	12



Source: Mertens and Bélanger (2010).

Large-scale plantations, notably of oil palm, have an enormous development potential in DRC. But so far, million-hectare investments in the Equateur and Oriental provinces by Chinese companies have not yet been implemented. Projects for planting fire wood have also been announced by private investors, but, so far, not on a very large scale and with limited outputs. Many observers expect the imminent development of large-scale industrial plantation agriculture and agribusiness in the country. This is the scenario favored by McKinsey in a report endorsed by the Government of DRC on the REDD+ strategy (MECNT, 2010). It foresees up to 2 million hectares of new forests converted into oil palm plantations before 2030.

The growing international demand for food, vegetable oils and biofuels has translated into growing pressure for controlling and developing new tracts of land worldwide. Agribusiness firms are already very active in Africa, where concerns on "land grabbing" are mounting. However, several observers point out the difficult investment climate in the DRC and the extremely poor state of the infrastructure and of the public institutions (Tollens, 2010). Land tenure issues, which involve a complex duality between modern law and local practices—where local chiefs can be powerful and often tend to behave as a landlord rather than a trustee—make land investments in this pluralistic land institutions environment risky. The limited capacities of the forest concessionaires to control the large surfaces that have been allocated to them sound as a warning for potential investors in large-scale plantations. The recent law on agriculture requires investors to associate with a Congolese citizen who should own no less than 51 % of the company share. Even though such specification has been often bypassed in several countries, the foreign private sector is highly concerned, especially since the law will not only apply to newcomers but also to those already operating. It recalls the "zaïrianisation" of the 1970s, which led to the collapse of the once flourishing agricultural sector.

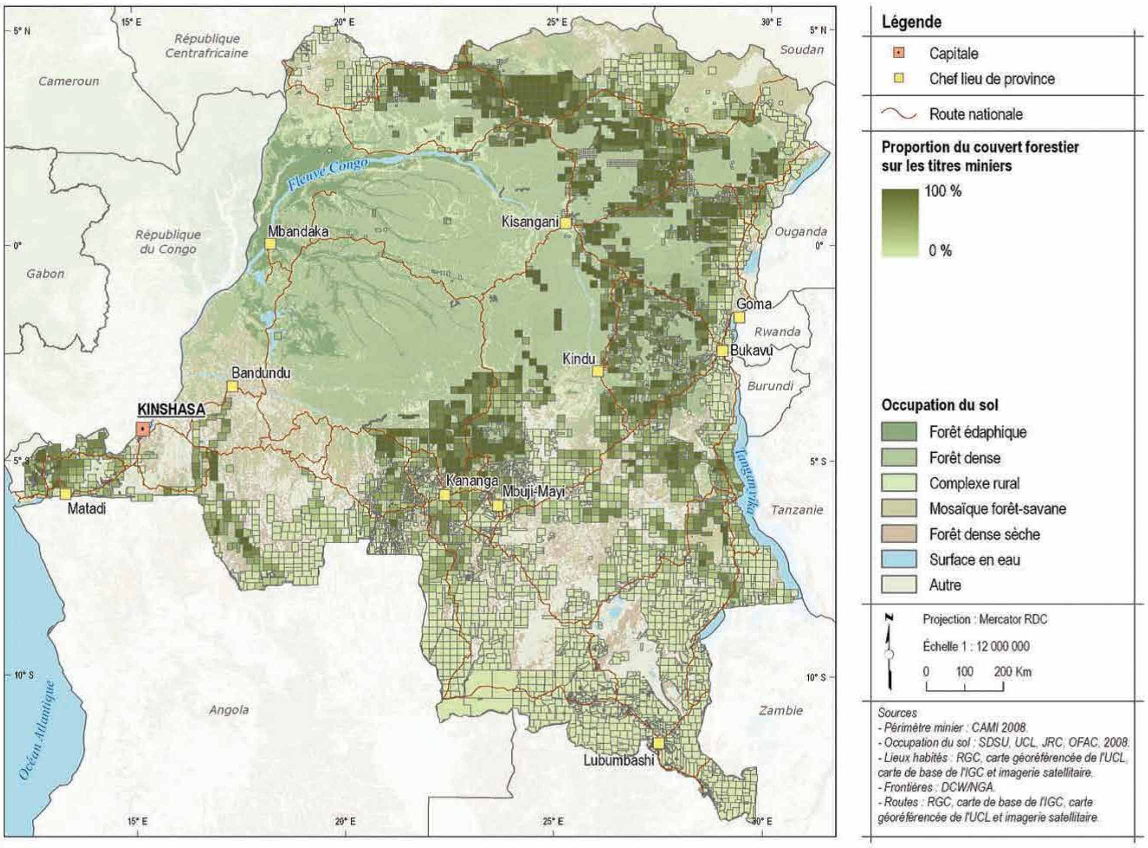
For all these reasons, predicting large changes in land-use in DRC is somehow uncertain, at least under the prevailing institutional and policy environment. Tollens (2010), a leading agricultural researcher on DRC's agriculture, sees the state of agriculture as "a declining and neglected smallholder agricultural sector, rapidly increasing food imports, and existing plantations trying to maintain only their productive capacity with replanting". Such statement contradicts Zhang *et al.* (2002), who predicted a rate of annual deforestation of 1.2% in 2030 due to the interlinked dynamics of population growth and shifting cultivation.

As for the large-scale plantations perennial crops, Tollens considers that "The investment climate and business conditions do not attract newcomers entering the sector. The lack of public support for the agricultural sector, the lack of adequate infrastructure and support services and Dutch disease type problems result in a lack of international competitiveness compared to similar forest areas in particularly South East Asia". Tollens adds that in forest areas "most young people prefer to migrate to the cities such that the population density will remain very low". Those diverging views on the dynamics of land-use are a big challenge for the setting of a REDD+ "reference scenario", which is meant to anticipate deforestation rates.

Mines and oil

DRC has one of the world's largest mineral mining and oil producing potentials. The allocation of "mining squares" by the ministry in charge of this sector is made without consideration of the land occupation (forest concessions or protected areas) and has a political priority, especially since the DRC does not have a permanent forest estate. Many mining permits (exploration or exploitation) have been allocated on forests, especially in the East. However, those permits are not systematically used and it is difficult to say if and when they will be used. Significant oil pools have been detected in the East and in the central "cuvette", where the forest cover is densest. Some national parks, such as the Virunga, are degraded by exploration and threatened by oil extraction.

FIGURE 17 Map of mining squares in forests.



Note: the greener the cell, the higher the proportion of forest covered by the mining square.
Source: Mertens and Bélanger (2010).

A dynamic REDD+ national coordination in a "fragile State"

The DRC has engaged in an active REDD+ preparation process, thanks to a dynamic REDD+ coordination unit that has launched numerous studies and initiatives, and is currently preparing the outline of what could be a cross-sectoral strategy. The DRC has benefited from significant financial support from foreign donors, and the ministry in charge of forestry and the environment has gained credibility. However, the ownership of this process, beyond a small community of stakeholders in Kinshasa and their international counterparts, remains an open issue. The 2011 agricultural code does not reflect REDD+ concerns, and land development ("*mise en valeur*") remains the compulsory condition for accessing better land tenure security. The absence of a permanent forest estate in the 2002 forestry code is now seen as impeding the national REDD+ strategy. The provincial authorities, who have gained power in the last decade, seem not to see any contradiction between continuing "business-as-usual" activities and getting remunerated through REDD+. The decree establishing community forests is still pending and it would probably give large powers to the customary chiefs, who in DRC have often shown a limited sense of accountability. Overall, the reach of the public authorities is very limited in a country that remains a "fragile State", where corruption is still omnipresent and the judiciary system is down (Karsenty and Ongolo, 2012).

3. *What do we know about the effectiveness of land management systems at the sectoral level?*

Market-based instruments: Promises not (yet?) fulfilled

An increasing volume of literature emphasizes the increasing use of market-based instruments (MBIs) as alternatives to administrative regulation and prescriptive laws for addressing environmental issues. In the 2009 TEEB document, the author states: “Experience shows that environmental goals may be reached more efficiently by market-based instruments than by regulation alone”. Before getting back to this statement, another issue has to be addressed to clarify the debate: are the so-called MBIs really about market(s)? For the same TEEB report, “Market-based instruments, such as taxes, charges or tradable permits can, if carefully designed and implemented, complement regulations by changing economic incentives, and therefore the behavior of private actors, when deciding upon resource use”. Put like this, one understands that MBIs are not necessarily about true markets but are encompassing a wide range of instruments that can modify the relative prices, hence create incentives for the economic agents. For Stavins (2005), MBIs are not outside the scope of regulation itself: “market-based instruments are regulations that encourage behavior through market signals rather than through explicit directives”. Incentives created through changes in relative prices seem to be critical in the definition, and – as the reference to taxes suggests in the TEEB report – markets are summoned as a metaphor, an organizing fiction of the world, for these incentives. In other words, MBIs use does not necessarily mean true markets and commoditization of nature’s elements – as we will see for PES – but are, first, about “achieving outcomes through the self-interest of the firms and individuals”.

Offsets

Environmental offsets schemes have been used for quite a long time in western countries. They are “voluntary or mandatory arrangements in which firms, industries or national governments offset unavoidable environmental damage in one location with investments in environmental conservation in another location (…) The Wetland Mitigation Banking operating in the United States is an advanced model of an offset scheme” (Swallow, 2007). In this family, one should mention the Transferable development rights (TDRs) system, a cap-and-trade mechanism by which forest holders (those who have at least an effective right of exclusion on the forest they use) can sell non-used development rights to other forest holders who need to clear the forest beyond the threshold (cap) they received. Such a mechanism needs setting a maximum of deforestation (cap) by zone and a stringent control mechanism. A mitigation banking institution can be set up to regulate the exchanges and reduce transaction costs. In Brazil, where deforestation is capped by law on the rural properties in the legal Amazon (20 % of the area in most cases), such a scheme has been implemented on pilot basis in some states and under the federal law, landholders have to replant land they cleared beyond legal limits or buy and preserve the same amount of forested land elsewhere to make up for what they cut. As pointed by Chomitz (2004), “Transferable development rights (TDR) programs offer a means of minimizing the opportunity costs (in foregone agricultural rents) of protecting a desired quantity of habitat”. However, in spite of the hopes placed in the spreading of such instrument, it seems its expansion has been impeded by several factors in Brazil. The main explanation is that its implementation lies on an effective enforcement of the law on the forest reserve, which has been barely done even though noticeable progresses have been made on this way under President Lula’s presidency that explains the positive results yielded by Brazil in curbing deforestation. Such a matter of fact recalls the necessary complementarities between MBIs and the enforcement of the rule of law. As with the international climate negotiation in which the incapacity of the international institutions to set a global emissions cap and to sanction those governments who do not fulfill their voluntary commitments has ruined Kyoto Protocol, the difficulty faced by the Brazilian government to properly monitor what is happening on rural properties and, moreover, to sanction non-compliers, represent a critical stumbling block for the use of TDRs in countries where the rule of law is barely enforced. In addition to that, TDRs, as many offsets schemes, need a well-established system of land property rights, a condition which is not fulfill in many developing countries.

Conservation easements

Given the difficulty to set and enforce cap-and-trade mechanisms in developing countries but also in many industrialized ones, another class of instruments is favored by conservation organizations, those which are setting “conservation easements” trough bilateral contracting with landowners or land users. Such bilateral agreements are widely used in North America, Australia and some European countries, and are exported in the developing world through the “conservation concessions” concept and some payments for environmental services (PES) schemes.

Under such common principle, the land owner or user receive payments from a third party to conserve all or part of the ecosystem it uses and have rights upon. It could be also, for a company, planting trees on the land owner estate, and acquire "carbon rights" that can used for offsetting carbon emissions, as it is practiced in Australia and New-Zealand. Conservation concessions have been designed, first, for turning industrial logging concessions into conservation areas in a context in which the forestland has legally to stay State's property. In Cameroon, such conservation concession scheme has been contemplated (for almost ten years now) by international NGOs to prevent the government to allocate an 830,000 ha area of primary forest to logging companies. But, here again, with some conservation concessions set in Guyana, Peru and, allegedly, in DR Congo, the expansion of such scheme has probably not matched the expectations of its promoters. This situation could however change if the REDD scheme were to become operational in the next future.

Payments for environmental services

Paying actors to conduct environmentally friendly initiatives or to give up destructive practices is the purpose of payments for environmental services (PES). Interest in PES has been increasing rapidly over the past decade. There are today more than 300 programs implemented worldwide predominantly used to address biodiversity, watershed services, carbon sequestration and landscape beauty. Nation-wide programs are run in China, Costa Rica, Ecuador, Mexico, Vietnam, the United Kingdom and the United States. Several empirical evidences tends to indicate that water-related PES have been more effective than others, probably because the payers are the direct beneficiaries of the service, unlike for biodiversity and carbon PES where the direct interest of the service's buyer – who act as an intermediary for the world's inhabitants and the future generations, to ensure additionality and absence of leakage outside the perimeter of the project is not so strong.

One of the most commonly used definitions is that of Sven Wunder (2005): "a voluntary transaction in which a well-defined environmental service (ES) or a form of land use likely to secure that service is bought by at least one ES buyer from a minimum of one ES provider, if and only if the provider continues to supply that service (conditionality)". This definition uses market terminology (buying), which is not free of ambiguities about the nature of the service that is the support of the transaction and can even create confusion about possible "ownership of the services" (in market relationships, one can only sell what one possesses). Environmental services are qualities associated with elements (for example the quality of water flowing through a drainage basin, or the carbon storage capacity of a forest) that are collective or public goods by nature. Furthermore, PES are not really about selling environmental services. In most cases, PES agreements provide for compensations for agreed restrictions on land use (e.g. stopping natural habitat destruction practices) and, in that sense, compare to conservation easements. Therefore, the amount of the PES differs from the monetary value of the service, just as in economics, the price is different from the value. If there is no market, as for biodiversity, the scope of the monetary evaluation is limited, especially as it is difficult to establish an economic value for heterogeneous assets. If the service has a market, however, as for carbon, the price of the service will depend on the relationship between supply and demand, but will not correspond to the market price due to operating and transaction costs.

The amount of a PES and the implementation of PES schemes therefore do **not** depend on the monetary evaluation of natural assets. They are determined by means of negotiations, which may or may not be balanced, and the amount should in principle cover at least the net cost of giving up an activity (the opportunity cost) linked to the usage restrictions or changes. Indexing payments on the opportunity cost nevertheless has certain disadvantages and negative side effects. "Carbon" PES (especially through avoided deforestation, the basis of the REDD mechanism) may be sources of financial gains for operators. In a carbon market (voluntary or regulated) with a single price per ton of CO_2 resulting from supply and demand, some agents providing an avoided deforestation service will have opportunity costs that are lower than the value of avoided emissions, calculated on the basis of the price per ton of CO_2. This difference between the "production cost" of avoided deforestation and its "purchasing price" creates a surplus. This surplus may be conserved by the agents, but will more likely be captured by carbon market brokers or PES project promoters, who will thereby pay themselves to varying extents. Moreover, conserving forests in agricultural frontiers in the Amazon instead of cultivating soybean, or in South Asia instead of planting oil palms, generates opportunity costs that are often high since these crops are very lucrative. PES programs will therefore concentrate on forests that are under less threat at the risk of paying actors who have nothing to lose by avoiding deforestation (zero opportunity cost).

PES are caught between two stumbling blocks: where the opportunity costs are high, the sums available are often not enough; but where the opportunity cost is low, the risk of paying for environmental services that are not endangered (lack of additionality) is high. Verifying additionality would require significant means in order to analyze local situations, which would imply higher costs. The Costa Rican PSA scheme is often considered a model, but has been criticized for not being sufficiently efficient (lack of additionality); Pfaff *et al* (2007) find very low impact of the PSA scheme on deforestation, since most of the payments went to landholders who would not have deforested even without payments. PES programs often make fixed uniform payments on a per hectare basis and have been criticized for that. The OECD (2010) pointed out that individual landholders are likely to have different opportunity costs of ecosystem service provision and suggests taking these differences into account. But such a choice encompasses other challenges. A major problem where PES and their social acceptability are concerned is that compensation based on the opportunity cost is inequitable for the poorest populations. Freezing user rights such as clearing, hunting or even the prospect of working in a forestry company deprives people of opportunities to lift themselves out of poverty. Moreover, within communities, it is often the poorest thatdepend on natural resources. By giving up certain activities, they lose vital access rights that are not generally offset by the payments, which are based on the average opportunity cost for the whole community. Nor is it unusual for these payments to be monopolized by the "elites". Simply compensating the opportunity cost for very poor farmers therefore raises ethical objections and is enough to justify envisaging another basis for payments.

Finally, adopting the opportunity cost as a basis for compensation does not prepare for the long term. Compensating for the loss of income from giving up certain subsistence activities may free up working time but does not release any new resources to acquire the capital needed to implement new agricultural or agroforestry technologies. Although a sophisticated national PES program in Mexico is based on assessment of various opportunity costs at local level, one can think that, in poorest countries, the feasibility of large-scale PES strategy will depend upon the design of "assets-building" PES, which means beyond the opportunity cost compensation logic. However, although the ecological intensification of agriculture is a necessary condition for reducing pressure on ecosystems, it is insufficient. This is seen in the mitigated results yielded by the Alternative to slash-and-burn (ASB) programs of the past two decades: with the extra income generated thanks to intensification programs, farmers developed new cash crops elsewhere at the expense of the forests - "rebound effect" (Fearnside, 1997). Hence the proposal to combine investment in more intensive agricultural technologies with direct incentives linked to ecosystem preservation provided by PES. Broader PES, in other words aimed at investment, may combine direct incentives with conditionality that was previously lacking.

Certification and eco-labeling

Forest certification has been probably the most successful MBI over the last two decades for improving producers' practices. Initially designed for tackling tropical forest degradation, and hopefully some deforestation threats by rising certified timber prices and makes forest conversion less attractive, it has become an unavoidable passport for woods willing to reach certain western markets. It is unclear if some certification schemes such as the PEFC (Program for the Endorsement of Forest Certification schemes) created in 1999, by the European timber industry for small forest owners, has really changed practices in temperate and boreal forest. As for the national certification schemes, such as the ones from Malaysia or Indonesia, endorsed by PEFC, their lack of independence vis-à-vis their governments has undermined their credibility for the buyers. The Forest Stewardship Council, launched in 1993, has gained a large audience and a relative credibility thanks to its independence vis-à-vis both the industry and the governments (Auld *et al*, 2008). More than 140 million were certified by mid-2010, with the bulk of surfaces concentrated in boreal and temperate region. One of the striking results in terms of FSC forest certification has been the unexpected high number of hectares certified in the Congo Basin, where governance is notoriously deficient. With around 5 million hectares (11 concessions) of exploited natural forests certified on three countries, the region can compare with the large Brazil where most of the certified areas are not logging areas. The impact of such certification is noticeable in companies' behavior compared to what it was before the first certificate was issued in 2005. According to Resources Extraction Monitoring, a specialized NGO having being appointed as watchdog to perform independent monitoring of forests operations in Cameroon, then in Congo-Brazzaville, certified companies are complying much more with legal requirements, which set quite high management standards that are barely enforced otherwise, than other enterprises (REM, 2010). Significant social achievements have been also reported in the large FSC-certified concessions. However, the cost of certifying large concessions is significant and the price premium brought by the certification is limited and likely to disappear when the market is turning down, as it has been the case in 2008 and 2009.

Forest certification under international schemes such as FSC provides some room for improvements of concessions management even within a context of law governance and public regulations failures, but such dynamic is likely to be restricted to a handful of companies exporting their products toward environmentally-concerned markets with few spin-offs on the other parts of the forest sector.

The FLEGT initiative

Illegal logging has been one of the issues ranking at the top of the forest-related agenda for around ten years. Several studies have suggested that illegal sourced timber from natural forests was exceeding the legal one in places such as Indonesia, DRC and the Brazilian Amazon discussed above (Scotland *et al.*, 2000; Djire, 2003; Lawson and McFaul, 2010).

The European Union has been engaged in bilateral negotiations with a certain number of countries exporting tropical timber to conclude voluntary partnership agreements (VPAs). These accords are intended eventually to prohibit entry into the European Union of timber from countries that have signed up to the VPAs but do not possess a FLEGT (Forest Law Enforcement, Government and Trade) licence to guarantee that the timber has come from a legal source. WTO rules would probably not allow the European Union to refuse entry to timber originating from countries that have not signed VPAs[5]. Negotiations are under way with Ghana, Indonesia, Malaysia and Cameroon and most of them have now signed up a VPA. Other countries (Gabon, Congo) should join these negotiations soon. Brazil has refused point blank to do so. Negotiations drag on with Indonesia and Malaysia because these countries are afraid that they will be penalised compared with competitor countries which reject the procedure (such as Brazil and China).

A recent Chatham House study (Lawson and MacFaul, 2010) finds that "while illegal logging remains a major problem, the impact of the response has been considerable. Illegal logging is estimated to have fallen during the last decade by 50 per cent in Cameroon, by between 50 and 75 per cent in the Brazilian Amazon, and by 75 per cent in Indonesia, while imports of illegally sourced wood to the seven consumer and processing countries studied are down 30 per cent from their peak". However, such estimations would need to be confirmed by reliable data, which do not exist since the bulk of illegal logging is associated with informal activities. Such small-scale logging activities supply domestic and, sometimes, sub-regional markets. As the timber produced this way is barely entering the international trade, it is essentially below the radar screen and extremely difficult to quantify without in-depth field studies. The measure adopted to exclude illegal sourced timber internal trade have certainly yielded significant results, but this does not means that quantities of illegal harvested timber have decreased. Several empirical evidences suggest that the duality in the forest sector in less-advanced tropical countries has widened in the two last decades.

Forestry within the climate regime: CDM and REDD+

Deforestation is a problem that mainly concerns developing countries. Yet these countries are not committed to quantified emissions reduction targets under the Kyoto Protocol. They only participate in the collective effort through the Clean Development Mechanism (CDM), for which tree planting projects are eligible. These are emissions reduction projects for which the promoters can earn certified "carbon credits", which are negotiable on specialized markets. To date, "forest" CDM projects (afforestation and reforestation) have been something of a failure: only 32 projects have been registered out of 3534 (at 18/10/11). The ban of forestry-CDM credits into the European Trading Scheme and the specific crediting regime for afforestation/reforestation projects ("expiring credits" valued less than the "permanent credits" allowable for energy-related projects) to take into account the risk of "non-permanence" of the carbon storage in forest biomass, have dissuaded many potential investors.

CDM is criticized by many as a poorly effective tool to tackle climate change. Being not a cap-and-trade instrument, its capacity to reduce true emissions lies on the "baseline scenario" which is a business-as-usual projection of the "without CDM incentives" situation.

Although framed by precise UNFCCC guidelines, the design of the baseline scenario is still a controversial exercise, not exempt from strategic behavior of the project proponent and the asymmetrical information (on the true marginal costs and benefits of the proposed activity) between the proponent and the appointed analyst opens the door to the crediting of many non-additional projects delivering "hot air" that are used subsequently as offsets for GES emissions. The verification system, by specialized companies, could create conflicts of interest and adverse selection: currently, third party verifiers are paid by project developers, with whom they do repeat business and thus are reluctant to contradict (Wara and Victor, 2008).

The CDM executive board has recognized loopholes into the additionality assessment and evoked a figure of 20% of non-additional projects. This figure is considered as understated by several observers and independent institutions (Schneider, 2008) who suggest figures beyond 40-50%. The instrument is also criticized for the potential perverse incentives it could generate in hosting countries: the CDM is said to encourage developing countries to keep their polluting industries (to get CDM credits to modernize them) and to lower national environmental standards for ensuring their CDM projects will remain "additional", i.e. go beyond legal requirement (Tirole, 2009). Finally, the extremely uneven distribution of CDM projects and benefits, concentrated mainly in China, India and Brazil, has cast doubt on the capacity of the instrument to genuinely contribute to the development of less advanced countries, and its credit has dropped in Africa where some have ironically renamed it "China's Development Mechanism".

From CDM to REDD

The "avoided deforestation" mechanism, acronymized successively as RED, REDD and REDD+, takes its roots in the debate on the eligibility of land use, land-use change and forestry (LULUCF) projects under the CDM, which was one of the most controversial issues at the Sixth Conference of the Parties in November 2000. One of the main reasons of the rejection of "forest conservation projects" from the CDM by a majority of the delegates was the risk of emissions leakage: without addressing the structural cause of deforestation, conservation projects are likely to simply displace deforestation pressure elsewhere, either directly or through changes in relative prices of crops and land (a constraint on additional arable land could raise crops prices and make therefore deforestation much profitable in other forests). The "compensated reductions" proposal came up in 2003 (Santilli *et al*., 2003) in the literature as a response to the "leakage objection" to the conservation projects in 2000: it was proposed that emissions abatement from deforestation would be calculated at nation's level, hence lowering – but not suppressing since there is also an international displacement of emissions from the LULUCF activities pointed out notably by Meyfroidt, Rudel and Lambin (2010) – the risk of leakage compared to a project-based approach.

Since 2005 and the proposal of the Coalition for Rainforest Nations, the REDD mechanism has been debated as a principle for remunerating developing countries that would reduce their deforestation rates. As debates progressed, however, the field of eligible activities was expanded due to pressure from different interest groups, both public and private. First, forest degradation, followed by forest management, tree plantations and finally the conservation of carbon stocks have entered REDD+ (as it has been known since 2007) one by one. Reducing degradation – which is particularly difficult to measure – was included to satisfy Central African countries which have low deforestation rates. Improving forest management would allow remunerating logging companies. Plantations, which are already included in the CDM, albeit under very strict conditions, were introduced by China which would like to see its industrial plantations subsidized, despite already being highly profitable. As for the conservation of carbon stocks, its meaning remains ambiguous: it may either refer to remunerating projects (as requested by large conservation NGOs) rather than states, or to compensating countries which have preserved their forests and want to be paid based on the amount of carbon that they still contain. The latter perspective has been bitterly defended by countries such as Gabon and the Republic of Congo, which promote the idea that the fact that their forested expanses are still largely intact thanks to the "virtue" of their public policies. Others would explain it through the absence of agro-industrial pressure and demand for land in these sparsely-populated countries.

REDD+: the end of a relative and fragile consensus

The continuous expansion of REDD+ to new types of activities is presented as making headway by those who confuse progress and rushing forward. In reality, each of these activities is subject to debate and has broken the relative initial consensus between three distinct interest networks – those who give priority to carbon, those who are concerned about biodiversity, and those who defend the rights of local and "indigenous" populations. To this one must add the frontal opposition between defenders of including REDD+ in carbon markets (payments would be made in the form of "carbon credits") and those who promote the idea of a global fund fed by an international tax system which has yet to be implemented.

Although, as mentioned above, the initial "avoided deforestation" proposal was not about project crediting, the project-based approach has come back in force in the debate and the field activities. Conservation NGOs and other forest carbon promoters look reluctantly at a national-based system in which they would depend of the goodwill of governments to receive carbon credits or direct remunerations. The world of business is also lobbying for REDD projects – which have begun flourishing everywhere in the tropics without waiting for an international agreement – to sell carbon credits on the market. It would not be very difficult, they argue, to convince markets that these projects are protecting forests "which would otherwise have been cleared". As for the CDM, the REDD+ mechanism rests on the creation of business-as-usual scenarios, which by definition are impossible to verify (if the project is implemented, then the reference scenario cannot be checked for validity), and thus subject to manipulation.

Moreover, rather than addressing the actual causes of deforestation, project-based approaches simply tend to displace the pressure of deforestation to other areas, which potentially cancels out the proclaimed carbon gains. To prevent such recurrent objection, the so-called "nested approach" (Pedroni *et al*., 2009) ambitions to maximize the advantage of earlier proposals for REDD+ payments, namely the combination of a national approach and a project-based remuneration. It allows for the local initiatives to sell carbon reductions generated by the projects in the global carbon market (or to be paid directly by an international fund) without interference from the government of the hosting country. Meanwhile, the other reduced emissions from deforestation or degradation monitored at national level and that are not attributable to projects performance will be attributed to the public action – and corresponding credits/financial rewards allocated to the government.

Despite the apparent advantages of the approach, it is not evident that such architecture is equally suitable in small (where all the forested area could be covered by REDD+ projects) and large countries: what if, at the end of a commitment period, all the REDD+ projects have shown emission reductions (and are credited for) while deforestation and degradation has increased at the national level? In other words, the real potential of such architecture to avoid leakages and opportunistic strategy of a government that would encourage, on one hand, REDD+ projects in some parts of the territory and, on the other hand, would foster land conversion (or is simply incapable to prevent it) in other parts, is questionable.

The political economy dimension of REDD+ is also often overlooked. This mechanism is founded on the hypothesis that developing countries 'pay' an opportunity cost to conserve their forests and would prefer other choices and convert their wooden lands to other uses. The basic idea is, therefore, to pay rents to these countries to compensate for the anticipated foregone revenues. The reference to the theory of incentives (in its principal-agent version) is implicit but clear. In this REDD-related framework, the Government is taken as any economic agent who behaves rationally i.e. taking decisions after comparing the relative prices associated to various alternatives, then deciding to take action and implementing effective measures to tackle deforestation and shift the nation-wide development path.

Such an approach ignores the very nature of the state, especially when dealing with "fragile" or even "failing" states facing chronicle institutional crises, which are often ruled by "governments with private agendas" fuelling corruption. Two assumptions underlying the REDD proposal are particularly critical: (i) the idea that the government of such a state is in a position to *make a decision* to shift its development pathway on the basis of a cost-benefit analysis that anticipates financial rewards; and (ii) the idea that, once such a decision has been made, the "fragile" state is capable, thanks to the financial rewards, to *implement and enforce the appropriate policies and measures* which could translate into deforestation reduction (Karsenty and Ongolo, 2012). The perspective of a "one-size-for-all" incentive instrument, which could be used without distinction in Brazil and in DR Congo, seems unlikely.

The way forward: what prospects for change?

The bleak perspectives surrounding the climate change negotiation seems to dismiss the perspective of an international regime designed around a global cap-and-trade system for curbing GES emissions in the next five years. Such a situation would have undoubtedly an impact on the use of MBIs, such as CDM and REDD+, and these perspectives are already reflected in the carbon price on the ETS market, lagging around €10 per ton of CO_2. The CDM shrank to $1.5 billion last year (2010) from $7.4 billion in 2007, according to World Bank estimates. Even if the EU moves ahead with the announced extension of its ETS scheme and its support to the CDM, uncertainty dominates in the compliance market, especially since new rules designed to better ensure the additionality of CDM projects are under preparation and could make more difficult projects registration in the future. In addition, the EU has decided not to accept any forestry credits, whether from the CDM or REDD+, as offsets for the ETS until 2020.

Instead, it seems that a fragmented regime is emerging, in which countries or bloc of countries will setup the rules of the game they intend to play for abating emissions. Emerging countries have refused so far to endorse quantitative cap on their emissions, even though China does not exclude to do so in the future. In the USA, the divide between Democrat Administration and the House of Representative dominated by the Republican Party makes unlikely the implementation of a cap-and-trade system once contemplated, and would make an agreement with the emerging countries in the international negotiation even more difficult. In face of shrinking compliance markets, the emergence of voluntary schemes and "over-the-counter" offsets is quite impressive.

The last State of Forest Carbon Markets states that "growing from already record-breaking years in 2008 and 2009, respondents reported a total of 30.1 [$MtCO_2e$] contracted across the primary and secondary markets in 2010. The estimated total value of transactions in 2010 was $178 million. The historical scale of the forest carbon markets climbed to 75 $MtCO_2e$, valued at an estimated $432 million with projects impacting more than 7.9 million hectares in 49 countries from every region of the world (...) The 2010 surge in the forest carbon market was fueled to a great extent by contracting from large Reduced Emissions from Deforestation and Forest Degradation (REDD) projects which supplied 19.5 $MtCO_2e$ out of the total 29.0 $MtCO_2e$ contracted in the primary market".

This voluntary forest carbon market raises however concerns about the possibility that a "carbon bubble" is emerging, fuelled by the demand of companies to become "carbon neutral" at least cost. Within this potential "carbon bubble", the REDD+ projects look particularly promising. Carbon traders have a lot to gain from a REDD+ credit bubble: companies or states which will buy these reductions to compensate their own emissions will happily purchase these inexpensive credits from project promoters without actually looking at the reality of the announced emissions reductions. The risk of creating an additional massive amount of hot air is very high. Companies are not alone in doing this: the state of California has already signed an agreement with the Brazilian state of Acre and the Mexican state of Chiapas to compensate part of its emissions through REDD projects.[6] Australian states are doing the same in Indonesia. A set of deregulated markets is filling the gap left by stagnant multilateral negotiations.

On the other hand, the growing importance of private governance instruments, such as certification and other voluntary schemes, such as the Roundtable processes (for protecting natural forest from soy bean or oil palm expansion) can continue to yield significant results. Roundtables processes are multi-stakeholder, long term and process-based approach involving a multitude of consumer and producing countries. The "Roundtable for responsible soy" seems to have yielded positive results in the Brazilian Amazon, and contributed to the shrinking deforestation observed since 2005 – even though the agricultural pressure has tended to move from the Amazon forests to the Cerrados, biodiversity and carbon-rich savannahs now endangered.[7]

FSC certification still has a potential of growth in the tropics, even though Greenpeace and other organizations try, especially in the Congo Basin, to prevent further certification of industrial logging on "intact forest landscape". The challenge, here, will be convincing the consumers and the governments of emerging countries, notably China and India, to pay attention to the impact if their consumption of both finished and intermediaries goods to the global ecosystems, and to favor eco-labeling in all their purchasing policies. Such a progressive change seems likely, but it could take time and being considerably delayed by the perception that it could hampers the impetuous economic growth which is seen by these governments as the only way to take out from poverty hundreds of millions people. A new trend in the Chinese economy, more oriented toward increasing citizen's welfare instead of the "full-export model" would help fostering changes in consumption patterns.

However, in producing countries, the increasing "informalization" of large fractions of the natural resources extraction economy, such as timber in Africa, reveals the limitations of such a MBI that rests on the purchasing power of concerned consumers.

The future of REDD+ is still extremely uncertain, at least as a united scheme. The architecture and the rules of governance of this instrument are still undecided and consensus seems extremely difficult to reach. Some market analysts argue that potential REDD+ credits would not be acceptable in a compliance carbon market as they embody unsolvable clearing problems for being traded on a derivative market (The Munden Project, 2011). The booming of REDD+ projects has probably more to do with the seizure of fleeting market opportunities under the fashionable REDD+ umbrella, and can hardly be considered as the pillars of a compliance regime that addresses the drivers of deforestation. So far, REDD+ is about public financing for the "readiness" phase, which is supposed paving the way to the market-based phase and the performance-based remunerations regime. However, the prospect for reaching this stage under a genuine compliance regime is more and more hypothetical, given the complexity of the so-called technical issues that barely hide diverging national and stakeholders' interests on the rules to be adopted. And without a global agreement on a new commitment period in the climate negotiations, the chances to see a market and performance-based REDD+ scheme in a regulated regime are even weaker.

But for the REDD+ process backed by public funding, the perspective can be different. Even in the current financial turmoil that impact the public finances of several industrialized countries, the need to get new financing sources to fill the deficits in national budgets but also to finance public goods find its way in the governments and public opinions. The striking rally of prominent European political deciders to the "Tobin Tax" (on financial transactions), which seemed very unlikely a couple of years ago, gives evidence of this new way of seeing aroused by the financial crisis. In the same vein, the implementation of "carbon taxes" on emissions, even on air and sea transport, is more and more considered in countries, along with cap-and-trade systems. A report commissioned by the United Nations on the Green Climate Fund intended to channel an annual US$100 billion in climate finance from the developed to the developing world (United Nations 2010). There is currently an on-going study commissioned by the G-20 on the effectiveness, revenue potential, and administration, of a wide range of fiscal options for climate finance. These include taxes on aviation fuels, maritime fuels, carbon, electricity, vehicles, financial transactions, as well as broader fiscal instruments (World Bank 2011a). If a portion of the expected money were earmarked to finance REDD+ policies, it could provide the needed means to address more effectively the drivers of deforestation and finance structural in-depth reforms that are needed to foster deep changes in the agriculture practices, the land tenure system and the land-use decisions processes. In such a public-driven REDD+ scheme, large-scale PES schemes, inspirited by those of Mexico or Costa Rica, could have a central role to provide incentives to local producers for a sustainable use of the ecosystems. The exact balance between such public-funded REDD+ dimension and the, mostly, private initiatives labeled "REDD+ projects" designed to yield carbon offsets is difficult to predict, but we can guess both will coexist.

4. *How are land use and land use change modeled in scenario exercises? Land use and land cover modeling and their assumptions*

Introduction

Anthropogenic land use/land cover (LUCC) change is non-random, yet it is hard to predict it accurately (Kaimowitz & Angelsen 1998) and prediction in the distant future is less accurate than prediction of the near future. Despite these challenges, the science of LUCC change forecasting has grown rapidly, thanks to the methodological advances in modeling the process of land conversion, which has led to the increasing need for an understanding of its impact and drivers. The rapid development of geospatial analytical tools and the easy availability of satellite data, which enable cheaper and more accurate analysis of regional and global LUCC[8], have also contributed to the rapid development of LUCC change models.

LUCC models have struggled with allocation land for competing land uses. Different assumptions and priorities have been used to predict land allocation. Such differences have led to different results, which underscore the disciplinary focus of modelers. For example, Rounsevel *et al* (2006) used a model which hierarchically allocated land in Europe as follows:

Protected areas > urban > cropland > grassland > bioenergy crops > commercial (unprotected) forest land > not actively managed.

This means that the modelers first took protected area as fixed and then hierarchically allocated land to the remaining land uses. Meanwhile, contraction of land in the priority land use (e.g. cropland) is allocated hierarchically from urban to commercial forest. The "not actively managed" land is regarded as unallocated (surplus land). This hierarchy could change according to the countries with food deficit. As argued earlier for example, first priority given to protected areas in food surplus countries may be given to food production in poor countries with food shortage.

Models for predicting future LUCC change use theory to link LUCC with its biophysical and socio-economic drivers discussed in the first section (drivers of LUCC). Biophysical characteristics such as temperature, rainfall and soil characteristics have an ecological influence on LUCC change. For example, compared to areas with low agricultural potential (e.g. arid areas), areas with high agricultural potential will need a smaller area to produce food for a given population. As discussed in the section on drivers of LUCC, socio-economic factors such as population pressure, income change, international trade and policies determine the demand for land-based products which in turn influence LUCC change. But since the future is not well-known, modelers have to make an assumption about the future stocks of land and trends of the LUCC drivers. In order to address the wide range of uncertainties over the influence and nature of some key drivers for LUCC, different scenarios also have to be used to run the models. Policy scenarios such as bioenergy mandates discussed earlier are used to guide LUCC modeling. Climate change scenarios are also used to explain LUCC.

Statistical approaches are used to establish historical relationship between LUCC and its drivers. Due to the historical data use approaches, the impact of drivers of LUCC differs significantly at different spatial modeling scales. Drivers of LUCC at global and regional scales are different from those at watershed or district scales. National or district models tend to incorporate more local level drivers of LUCC, while global or regional focus on broader regional and global drivers. Due to the global nature of this study, we restrict our review to regional and global models. Regional and global LUCC models are still few and largely focus on the influence of drivers such as climate change and water availability on the change in the agricultural productivity and cultivation patterns, and its impacts on key dimensions of environmental quality such as of biodiversity, climate change, and agriculture and water outlook (Fischer and O'Neill 2005).

Annex 1 discusses the major LUCC models and gives some key characteristics which guide their assumptions and scenarios. Annex 1 also gives the history of development of the models along scientific disciplines and how over time they have tended to be more integrated. The table below summarizes the discussion in Annex 1 by giving the strengths and weaknesses of each model.

TABLE 4 Summary of LUCC models.

Type of model	Example[a]	Major strength(s)	Major weakness(es)
Geographic models			
Statistical	CLUE	Uses cause-effect regression approaches to project LUCC change. Historical data of the drivers of LUCC and LUCC change are used.	Use of historical data may not fully reflect unexpected trends and patterns.
Rule-based	SALU	Captures land intensification and combines spatially explicit quantitative approaches with qualitative approaches (fuzzy logic).	Use of qualitative approaches may not allow interpolation of result and could be prone to subjective judgment.
Economic models			
Partial equilibrium models (PEM)	IMPACT, FASOM	Considers one of few sectors – thus able to include greater details of drivers of LUCC change of the selected sectors.	Assumes away the feedback/effect of excluded sectors on the included sectors.
General Equilibrium models (CGE)	GTAP, GTAP-AEZ, IMAGE,	Considers all sectors – thus incorporates feedback of all sectors.	To be tractable, CGE models don't include details of one sector – hence losing rigor of individual sectors.
Integrated PEM & CGE models			
	GLOBIOM, MAgPIE	Takes advantage of PEM and CGE models.	Acquisition of data for all sectors may be a problem. Convergence may become a challenge if a lot of variables are included in each sector.
Ecological models			
	SDM, SAR	Broader consideration of ecosystem services – including non-tradables.	May be hard to put value on some ecosystem services.
Integrated geographic, economic & ecological models			
	Patuxent Landscape; General ecosystem model	Takes advantage of all advantages of models included.	Acquisition of data for all components may be a challenge. Convergence and rigor of subcomponents is a challenge.

[a]Please see annex 1 for references and other details of the models.
Source: Authors' compilation.

Assumptions of land allocation & drivers

Over time, LUCC models have tended to realize the impact of both biophysical and socio-economic drivers of LUCC. Yet, there remains a strong bias towards the orientation of the models to the different disciplinary branches out of which they originated. In the discussion that follows, we will consider 3 types of models: those that are mainly driven by economic market-equilibrium principles; those that focus on geographic criteria for land allocation; and those that concentrate more heavily on ecological impact as criteria for managing land cover change. Economic models continue to focus on socio-economic drivers and give limited consideration of biophysical characteristics. Due to this focus, the economic models have more effectively reflected international trade and globalization and how they affect LUCC. They have captured better the influence of policies and other socio-economic factors on LUCC. Additionally, environmental and natural resource economic models have continuously incorporated biophysical characteristics in the land allocation and its impacts on ecological services. As such they have been solving the economic models by exogenously incorporating the biophysical characteristics and incorporating them in the economic models as state variables. A good example is Evans *et al* (2011) who used species distribution models (SDM) to allocate land to biofuel production in US such that biofuel production had minimum impact on biodiversity. However, many economic models still remain weak on their analysis of land allocation. This leads to allocation of unused land to agriculture or planted forests – in order to account for the increasing demand. Even though recent concern on GHG emission have prompted many economic modelers to consider impact of LUCC on GHG, few economic models take into account externalities of land allocation on ecological services. This leads to allocation of fragile land areas to anthropogenic land use in many of the *ex ante* projections that are simulated with these models.

TABLE 5 Major assumptions of major LUCC models.

Assumptions	Type of LUCC models		
	Economic	Geographic	Ecological
Ricardian allocation – international trade drives global land use	***	*	*
Population pressure impacts conventional ecosystems[a]	***	***	***
Income increases demand for food & other land-based goods and services	***	**	*
Own and cross price elasticity of demand and supply drive land use	***		
Bioenergy policies & mandates increase land conversion	***	**	**
Zoning (land allocation) is driven by land suitability and species abundance[b]	**	***	***
Land allocation protects biodiversity and ecosystem services	*	**	***

Asterisks indicate strength of focus: *** =strong; ** = medium; * = weak.

[a] Conventional ecosystems are ecosystems with minimum or no human influence.

[b] Meyfreidt and Lambin (2011).

Source: Authors' compilation.

Geographic models continue to be more focused on the allocation of land based on the suitability of land use and the spatial location of ecosystems and population. This has led to geographic models that better allocate land use to areas with minimal effect on the ecosystems. The models better capture the potential productivity of different land uses and are better able to reflect land management than economic models. As discussed above, however, geographic models assume that prices and other international feedback variables are exogenous (Schneider *et al*., 2011). This makes them less able to reflect the influence of international trade on market-driven agent behavior, which has reduced the strong correlation between population density and land use that is observed in historical data.

Ecological models have linked land allocation to species abundance and extinction, ecological footprints and other environmental concerns. Ecological modeling methods also assume, largely that prices and other economic variables are exogenous factors, thus failing to fully account for their impacts and trade-offs required in land allocation. Table 5 summarizes the assumptions of the major LUCC model types and shows the focus of each category. Below, we assess how realistic each of these models and the reasons behind their differences.

Factors contributing to disagreement of land allocation across LUCC models

There are many reasons leading to the disconnect among different models. We highlight the major drivers behind such a disconnect:

(i) *Disciplinary focus*, which does not take into account influence of other factors outside researchers discipline. To conduct rigorous analysis, researchers tend to focus on their scientific discipline and ignore drivers of LUCC, which fall outside their discipline. Perhaps this is the foremost reason for the disconnect across LUCC models.

For example, traditional LUCC economic models have estimated land owners response to land policies using standard economic utility functions accounting for only marketed goods and services and ignoring the influence of non-pecuniary benefits/costs such as cultural (e.g. esthetic, heritage) and regulating and support services (Newell and Stavins 2000; Plantinga 1997). This leads to models that do not do a good job of predicting the actual landowner response. Likewise, traditional economic LUCC models tend to ignore biophysical characteristics, thereby failing to account for one of the most fundamental drivers of LUCC. On the other hand, traditional biophysical models ignore socio-economic factors, which, as this report has already shown, play a big role. For example, the SDM LUCC models tend to only take into account human population and ignore the traditional socio-economic characteristics such as policy mandates, prices and elasticities.

(ii) *Scenarios, assumptions and model structure used.* Even for cases in the same scientific discipline, disconnect among results is common. We illustrate this important point by examining economic models that analyze the impact of biofuel mandates on LUCC. We recognize that the results of the quantitative assessment depend on some key modeling assumptions, which differ across different categories of economic models, such as assumptions on yield potential on newly cultivated land, the responsiveness of yield to price changes, the spatial representation of trade and other issues (see Witzke *et al*, 2010, Edwards *et al*, 2010). Much of the debate over the indirect LUCC (iLUCC) effect of biofuels, in terms of its size and specificity to particular biofuels production pathways, relates to some of the quantitative uncertainties in the modeling, and speaks to the need to think through and guide the use of quantitative (and even qualitative) impact assessment techniques within the policy design process itself (Nassar, 2011). Some research efforts have been underway to try and understand the differences between models and types of iLUCC effects that they generate, and we summarize a set of results, as an illustration of this, in Table 6 below. These results are drawn from a recent comparison study by Witzke *et al*. (2010), which tried to subject various models to biofuels shocks in order to illustrate (and understand) the differences in impacts on land use change among them. Among the many factors that underlie these differences are those of basic model structure, since some of the models are partial-equilibrium in nature (like the IMPACT, AgLink, FAPRI models) and focus mainly on agricultural markets and consumption, whereas other models take all interactions within the economy into account (like GTAP and LEITAP) and bring all markets (including input markets for labor, capital and chemical inputs) into equilibrium with respect to the behavior of the agents within the economy. Some differences come from the way in which the by-products of biofuels are handled, which offsets the decrease in feed demand when grain or oilseeds are used for biofuel feedstock production. Other differences come from the variation of parameter values used for key behavior relationships, such as the response of area or crop yield to price, which differ according to the particular form of the functional relationship that is embedded in the model (linear versus non-linear, etc.). Differences in how models handle trade also affect these results – as some models have a detailed bilateral representation of trade flows, such as in the GTAP models, versus a 'pooling' of total net trade from all countries within an integrated world market, as is the case with many partial-equilibrium models. Indeed, there is a constellation of possible influences that could lead to these differences, which have been discussed in more detail by other authors (Edwards *et al.*, 2010; Nassar *et al*. 2011).

TABLE 6 LUCC effects in different models.

Land use change (ha/ton biofuels)	US ethanol		EU ethanol	
	maize	wheat	coarse grains	wheat
IMPACT model	0.12	0.22	0.29	0.22
AgLink model	0.51			0.57
FAPRI model				0.39
GTAP model	0.16			0.79
LEITAP model	0.86			

Source: Witzke *et al*. (2010).

(iii) *Data used.* There has been improvement in data capture and quality over time (Hansen *et al*. 2008) due to increasing geo-spatial technology and new satellite systems (e.g., Landsat, MODIS, China-Brazil Earth Resources Satellite [CBERS]).Yet, data availability, quality and nomenclature remain key challenge for LUCC model results. Large inconsistencies exist across data sources. This leads to different results even for models using the same assumptions and approach. For example, the Advanced Very High Resolution Radiometer (AVHRR) data (10km resolution), states that irrigated area in India in 2000 was 132 million ha while the 0.5 km resolution Moderate Resolution Imaging Spectroradiometer (MODIS) data states the area to be 146 million (Thenkabail *et al* 2009),a 10% difference. Using average global cropland and forest area from different satellite data sources as a baseline, Fritz *et al* (2011) compared global cropland and forest area derived from GlobCover, MODIS v.5, and GLC-2000 and found a differences ranging from 23% to 36% for cropland and 8% to 18% for forest areas (Table 7). The major reasons for the inconsistencies include classification methodology, training and ground reference data differences, satellite sensors used and georeferencing errors. Consistent with Aspinall (2004), difference in dates of data collection also contributed to the inconsistencies.Attempts to harmonize land use classification have been made and some methods of calibration of different data sources have been developed (Ibid). Remote sensing data sets are often the start of more derived data inputs, such as historical reconstructions of land use. These reconstructions use satellite data to disaggregate agricultural census and survey data and represent land uses at the pixel level. Many of these products are excellent for global modeling efforts but rarely have the specificity needed at regional scales. The selection of input data sets is left to the modeler, who may select based on their interpretation of the best data set for their study. New methods of comparing among land use data sets are giving rise to new hybrid data sets that weight the most likely land covers and give total land use (e.g., cropland) as a percent probability (See Geo-wiki.org for more details). Additionally, groups are using satellite data sets to derive regional LUCC for input to models.

TABLE 7 Disagreement of global land cover data.

Land cover products compared	Cropland (Mha)	% Disagreement*	Forest (Mha)	% Disagreement*
GLC-2000 vs MODIS v.5	325.8	23.4	730.8	**18.5**
GlobCover vs MODIS v.5	505.9	36.4	387.2	**9.8**
GLC-2000 vs GlobCover	395.2	28.4	314.3	**8.0**

* The reference figure to which the LUCC models are compared is the average value of all three LUCC models.

Source: Fritz *et al* (2011).

Accuracy of prediction of LUCC models

Prediction of future LUCC remains elusive since the future trends and pattern may not be known. In fact only few validation studies have been done (Verburg *et al* 2004; Kok *et al* 2001; Rounsevell *et al* 2006; McCarlla and Revoredo 2001). One major challenge is the distant future that models predict, making it harder to evaluate their accuracy. For example, most current models have been predicting the future in 2030 and 2050 – making it harder to assess their accuracy today. In many cases, choice of such distant future is dictated by the speed at which some systems change and/or evolve. For example climate change scenarios use distant future due to the slow change that may not be significant in a decade or two. We first compare the LUCC model projection and actual land use change and then investigate the reasons for differences.

Comparison of selected model projection and actual LUCC

As an illustration, we use five models that project food production and/or consumption. The differences between the projection and actual food production is proportional to the corresponding differences in land use change since the models use supply equations in which land is included as one of the driver of food production. To simplify the comparison of accuracy, the selected models projected food production and consumption to year 2000 from a baseline of around 1993-1997. Table 8 shows that all models overestimated global food production. The GOL model projection had the smallest error while the FAO and IFPTSIM models had the largest errors. Consistent with Pontius *et al*. (2008) but contrary to Verburg *et al.* (2008), McCarlla and Revoredo (2001) observe that the global models tended to cancel errors encountered at regional level and therefore tended to be more accurate than corresponding regional models. For example, the World Bank projection model overestimated global food production by about 6% but overestimated food production in SSA and East Europe and former USSR by over 40%.

TABLE 8 LUCC model projection versus actual food production and consumption in year 2000.

	Region	FAO	IFPRI	World Bank[d]		USDA
		World Food Model[a]	IFPTSIM[b]	IMPACT[c]		GOL[e]
	% Projection error (+ is overestimation, - is underestimation)[f]					
World	Production	8.9	8.9	7.3	5.9	2.8
	Consumption				**4.5**	**2.6**
Production	High income countries	10.8	20	9.9	8.2	5.9
	Eastern Europe & former USSR	54	67.1	45.4	47.6	20.7
	Low & medium income countries	7.3	−0.3	5.1	4	0.3
	LAC	3.8	−2.5	−5.7	−4.8	−2.4
	SSA	8.1	−3.2	1.4	41.3	8
	Asia	4.3	−3.1	2.2	−0.9	0.3
	MENA	44.5	36.1	44.7	33	−2.3
Consumption	High income countries				2.8	1.6
	Eastern Europe and former USSR				43.9	20.7
	Low & medium income countries				5.6	3.3
	LAC				−4.9	−4.8
	SSA				25	1.6
	Asia				3.5	9.3
	MENA				17.3	−20.4

Notes: Regions: LAC=Latin American countries; SSA=sub-Saharan Africa; MENA=Middle East and North Africa.

Models: GOL = Grains, Oilseed and Livestock Model; IMPACT=International Model for Policy Analysis of Agricultural Commodities and Trade; IFPTSIM=International Food Policy & Trade SIMulation.

[f]Blank means data not available or not calculated.

Sources: [a]FAO (1994); [b]Agcaoli and Rosegrant (1995); [c]Rosegrant (2001); [d]Mitchell, *et al* (1997); [e]USDA (1997) – all as cited by McCarlla and Revoredo (2001).

There have been a number of studies whose predictions have proven to be very different from reality. A classic example of predicting the future while relying on assumptions that are entirely based on historical data is the famous prediction made by Malthus (1888) that periods of future mass starvation were inevitable due to the assumed inability of the earth to provide enough additional food for the large projected increases in population. More recently, other authors have also predicted the doomsday out of fear of the rapid global population growth. Paddock and Paddock (1967) predicted a massive starvation in 1975. Similarly, Ehrlich (1970) predicted a population "bomb" – a rapid population increase which would overwhelm food supply and lead to mass starvation.[9]

Buringh (1985) predicted that highly productive land will all be converted to agriculture by 2000 but contrary to this, 31% of land suitable for agriculture was still available in 2009 (Bruinsma 2009). There have also been optimistic predictions of LUCC and its consequences. Clark (1970) and Brown (1967) concluded that the world will be able to feed itself due to technology development, a conclusion which is consistent with Boserup's (1965) induced innovation theory in which agricultural intensification and innovations occur in response to increasing population and consequent higher land value.

Reasons behind discrepancy between LUCC projections and actual land use change

Overall, accuracy of prediction is determined by a number of factors. We explore the major reasons below.

Technological change: Malthus failed to take into account technological progress, which has averted the doomsday scenario. Surprisingly, many current socio-economic modeling approaches still fail to incorporate agricultural or other land use intensification—improvement of land management practices (Lambin 2001). Part of the reason behind this is the difficult of predicting new technologies and incorporating them in simulation scenarios (Ewert *et al* 2005). Even when incorporated, assumptions about future technologies are subjective (McCarlla and Revoredo, 2001).

Failure to include human behavior and policies: Future projection of land use is also limited by failure to include human behavior and country specific policies, both of which are major drivers of LUCC. Models including human behavior require a short-time span, typically of 10-20 years to make long-term projection (Heistermann *et al* 2006). The big challenge is inclusion of policies, which may continually change with the electoral cycles and shifting perceptions of political opportunity or risk. A good example is the Amazon region, which a number of predictions have been made that it will be wiped out. Inclusion of human behavior and policies may also not be feasible for regional and global models due to variability, which makes it hard to generalize them at larger spatial area.

Data spatial resolution and variance: Holding all else constant, accuracy of the regional and global LUCC models increases with spatial resolution. Verburg *et al* (2008) showed that in Europe, high spatial resolution models allowed incorporation of local drivers of LUCC in the modeling and this improved accuracy of LUCC prediction. However, such models are data intensive and may not be incorporated in regional and global models including regions with data dearth – e.g. SSA. Accuracy of LUCC models at smaller spatial scale may be different. A review by Pontius *et al* (2008) showed that the accuracy sub-national model predictions was greater when coarser resolution data were used. Possible explanation for this puzzling result is that coarse resolution resolves the conflicts of the finer pixels (Pontius *et al* 2008).

Data with greater variability has higher forecasting accuracy since they better capture the signals and relationship of LUCC and its drivers. A review by Pontius *et al* (2008) of nine LUCC sub-national models concluded that prediction of models was more accurate for models using baseline data showing greater variability. The Pontius study also concluded that the study site, time and data format play a big role in model accuracy. Aspinall (2004) also concluded that performance of models varies over time, and this poses a challenge in validation and calibration of models. Even the same model may produce quite different results when used in a different site, time and/or data format (Pontius *et al* 2008).

Unexpected events: Some unexpected events can drastically change expected LUCC change patterns and trends. Recent trends which have surprised LUCC modelers include the following.

(i) *Amazon deforestation trend.* As discussed earlier, deforestation of the Brazilian Amazon was reduced by 74% in 2009 compared to its level in 2003-2004 (CBD 2010), a reduction which has surprised modelers. This has led to some optimistic outlook predicting "the end to deforestation" (Nepstad *et al.*, 2010). Under their model, they projected deforestation under business as usual to reach about 28,000 km^3 per year but actual deforestation was only about 8,000km^3 (Ibid). Such large difference is due to the difficulty in predicting a bold decision, which Brazil took in 2008 to end deforestation and the consequent commitment by the international community to support such decision (Ibid). [10]

(ii) *"Land grabbing" and high food prices.* Foreign land acquisition increased rapidly following the food price spike in 2005/06 (Figure 8). As shown on Table 3, between 3% to 10% of forest and pasture land area may be converted to cropland for biofuel production. This has defied the long-term downward trend in food prices, which had persisted since the 1960s. Models which used those historical trends did not capture the high food prices and apparent price volatility which prevailed in 2006-2008 and 2009-2010 (FAO 2011b). For example, the IMPACT model predicted decreasing food prices to 2020 and improving food security (Cassman 2001). The foreign land acquisitions discussed earlier also surprised many modelers and this has led to reevaluation of the food price patterns in LUCC models.

(iii) *Weakened population pressure-land conversion relationship at local level.* Past LUCC models have predicted land conversion to anthropogenic ecosystems as population pressure increases. However, recent empirical evidence shows a more complex relationship. A global study by Bai *et al*. (2008) showed a positive correlation between change in population density and greenness, measured as normalized differenced vegetation index (NDVI), which measures density, condition, and the health of photosynthetically active plants. However, regional and country level analysis has shown that such pattern is not consistent throughout. In countries with strong government effectiveness, change in population density was positively correlated with NDVI and the contrary was true for countries with weak government effectiveness (Nkonya *et al* 2011). Even within countries there is complexity in the relationship of population and deforestation. DeFries et al. 2010 used satellite estimates of forest loss and found that within many countries deforestation is positively correlated to urban population growth and agricultural exports. As noted earlier, international trade has also weakened the local population pressure-land conversion relationship.

5. Prospects for the future

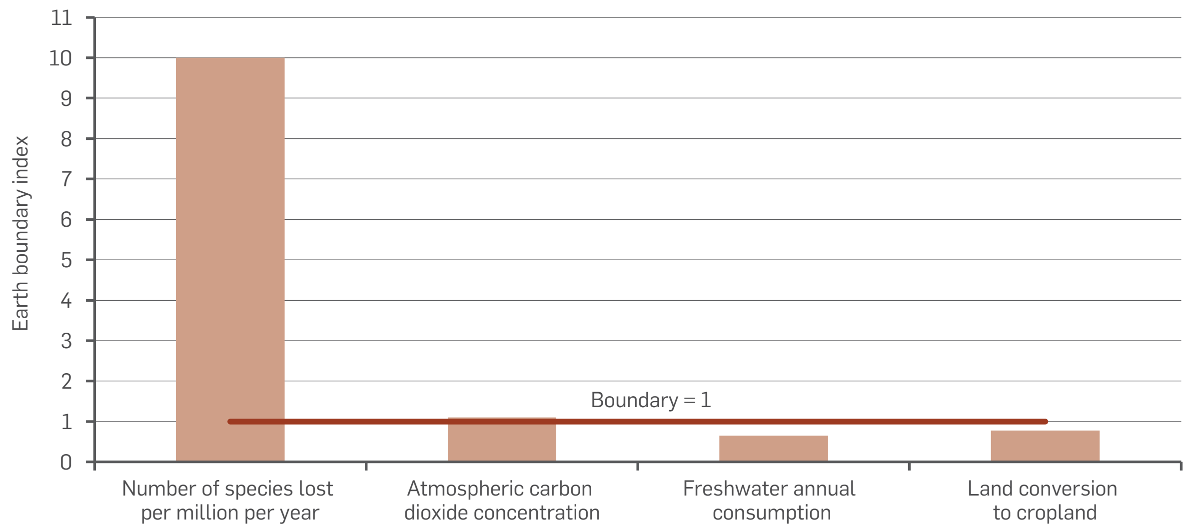
Given past trends, what can be expected from a business as usual scenario?

The analysis above shows the conflicting demands for land use, which are unsustainable under business as usual (BAU) scenario. We examine the impact of LUCC change on key earth systems to illustrate what would likely happen under BAU.

Land use conversion: As argued throughout this paper, LUCC change is posing a grave danger to earth's ecosystems. As discussed above, per capita arable land area is decreasing fast. Rockström *et al*. (2009) estimate that the safe upper boundary of the global cropland area is 15%, a level that is only about three percentage point of the current cropland of about 12% (Figure 18). However, the UNFCCC Commission on Sustainable Agriculture and Climate Change[11] has concluded that the current global agricultural production has already stepped outside the safe boundary (i.e. maximum amount of food that could be produced under a given climate to provide minimum food requirement of a growing population with minimum impact on the climate) (Beddington *et al* 2011). In addition to its effect on climate change, the conversion of forest, wetlands and savannas into agricultural land is not sustainable as this reduces the ecosystems capacity to provide regulating services and biodiversity (Elmqvist *et al* 2011; WBGU 2011; Rockström *et al* 2009). As will be argued below, sustainable intensification is the only option for achieving food security.

Biodiversity: Recent efforts to increase forest cover through reforestation and afforestation programs have helped to reduce deforestation but they do not fully restore lost biodiversity, which is built over hundreds of years and comprises complex and diverse biomes. Biodiversity trends monitored using the living planet index (LPI) show that since 1970, biodiversity has declined by 30%. The tropics have had a severe decline in biodiversity (about 60%) whereas the temperate regions experienced relative recovery (increase of +15%) (CBD2010). Rockström *et al* (2009) also report that, on average, more than 100 per million species are lost each year (E/MSY)–a level that is more than 100 times the planetary boundary (10 E/MSY) deemed to be safe operating space for human welfare within the earth system. Current rate of extinction is 100-1000 higher than the Holocene (pre-industrial) age level (0.1 – 1 E/MSY)(Figure 18). BAU is not an option as it has already proven to be unsustainable.

FIGURE 18 Global boundaries versus current status/use of biodiversity, CO_2 emission and freshwater consumption.



Earth system	Indicator	Current status	Boundary
Biodiversity loss	Species extinction/million species/year	>100	10
Climate change	Atmospheric CO_2 concentration (ppm)	387	350
Freshwater use	Global consumption/year (billion cubic meters)	2600	4000
Cropland area	% of global land area	11.7	15

Source: Calculated from Rockström *et al*. (2009).

Freshwater resources: With a rise in population, there is an increase in the quantity of water required for agricultural production, domestic consumption, industrial use and recreation. Currently, about 17% of the 7 billion people experience severe water scarcity (FAO 2011c). Over the past 50 years, freshwater withdrawal tripled (UN-water 2011) while irrigated area increased 117% (FAO 2011a). During the same period, rainfed crop area decreased by 0.2% (FAO 2011). Groundwater is increasingly becoming a major source of irrigation water; by 2009, groundwater accounted for 40% of the volume of irrigation water (Ibid). This is leading to falling water tables and puts at risk the inland arid lands of India, China, the Midwestern United States and the MENA region, which heavily depend on groundwater for irrigation (FAO 2011). Climate change, water pollution and land degradation are all increasing the uncertainties of freshwater resources, further putting pressure on the available freshwater resources. The situation is more alarming in arid areas in developing countries, which experience severe water shortages.

Bioenergy: Demand for energy will increase 35% by 2035 compared to its 2008 level (IAEA 2010). Bioenergy production has been one solution to addressing this rising demand. Half of the global cereals consumption in 2005/6-2007/8 was due to US ethanol production (Hertel 2011) and projections by FAO/OECD (2008) show that 52% of maize and 32% of oilseeds demand to year 2020 will be due to bioenergy. Estimates show that a large portion of the area for bioenergy production will be derived from clearing forests and grassland (Lambin 2010; Hertel 2011). This trend shows the trade-offs between the global objective of reducing GHG emission and biodiversity since the conversion of forest and grassland to bioenergy reduces biodiversity. At the same time, switching cropland used for food production to bioenergy production will lead to higher food prices, which in turn will compromise the global objective of eradicating hunger by 2015. For example, Hertel *et al* (2008) estimate that EU and US biofuel mandates will reduce pastureland in Brazil by about 10% in 2015 from its 2006 level. How much this reduction in pastureland will affect other land uses, however, depends on the extent to which stocking densities of livestock are likely to change (Dumortier *et al* 2010), which is often poorly captured by many models of agriculture and land use. The discussion above suggests the uncertainties of reducing GHG emission using the BAU (first generation) feedstock. Consideration for the second generation feedstock have been argued to be a better option for achieving the environmental objective of reducing GHG.

What is achievable?

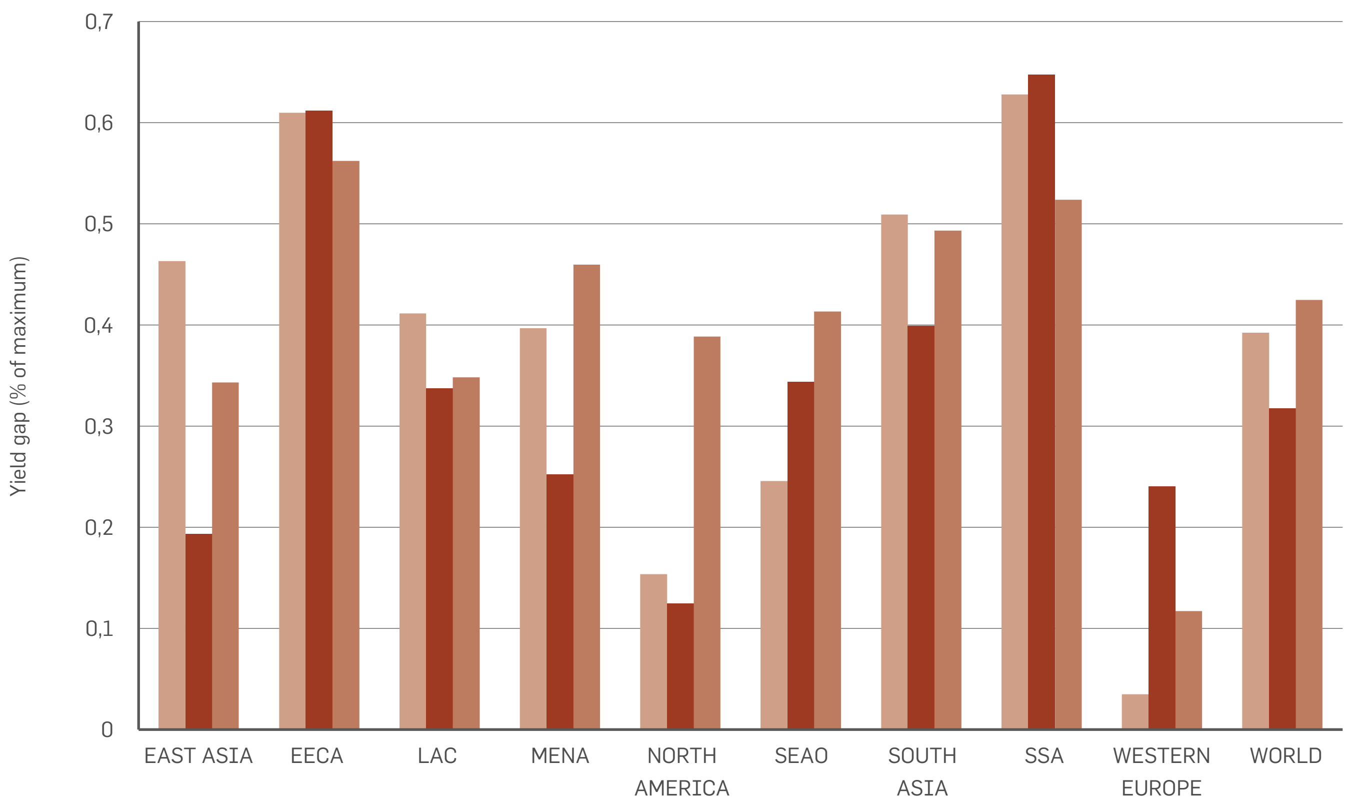
We explore the prospects for various scenarios and how realistic the assumptions used in international debates are. We focus our discussion on food security, climate change and biodiversity. Both food security and climate change were not a focus of our discussion on LUCC but they both have been dominating international debate on sustainable development. The focus on food security is based on the fact that agriculture contributes the largest share of land conversion and that recent food price spikes have renewed debates on food security (Fan and Pandya-Lorch 2012). Focus on climate change is largely based on the international debate on mitigation– an aspect which has dominated efforts to increase forest area and forest conservation efforts.

Food security

The Millennium Development Goals state that by 2015, the share of the people with hunger will be reduced by 50% from its level in 1990 (MDG 2010). This goal has seen limited achievement in developing regions, where the proportion of people suffering from undernourishment was 20% in 1990-92 but fell to only 16% in 2005-07 (MDG 2010). One of the strategies for addressing hunger is to increase agricultural productivity in developing countries. Sustainable agricultural intensification will require adoption of sustainable land and water management practices (FAO 2011a). This includes use of more efficient land management practices and irrigation water (Ibid). Increasing agricultural productivity will be more significant in developing countries where there is still a wide yield gap. Figure 19 shows the yield gaps of the regions.

A recent forecasting study showed a decreasing yield growth at global level (Figure 20). The major reason behind the downward trend is the narrowing yield gap in developed countries and major producers in Asia. This means developing regions with wide yield gaps will account for the largest share of production growth to meet the future increase demand. This is achievable but to realize this, constraints which limit higher yield in such regions need to be addressed. These include increased investment in agricultural research as well as addressing market conditions and rural services, which will provide technical support and incentives for increasing productivity. Greater water productivity (Falkenmark and Rockström, 2006) – is also required to increase yield in the regions where water productivity is low.

FIGURE 19 Yield gaps of major food crops across regions.

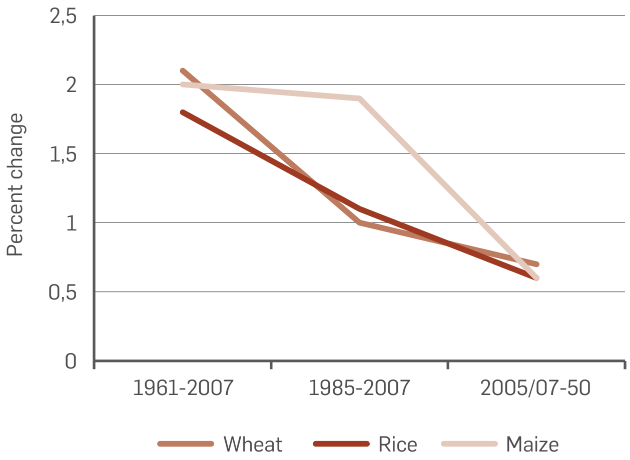


	EAST ASIA	EECA	LAC	MENA	NORTH AMERICA	SEAO	SOUTH ASIA	SSA	WESTERN EUROPE	WORLD
Maize	0,46	0,61	0,41	0,40	0,15	0,25	0,51	0,63	0,03	0,39
Rice	0,19	0,61	0,34	0,25	0,12	0,34	0,40	0,65	0,24	0,32
Wheat	0,34	0,56	0,35	0,46	0,39	0,41	0,49	0,52	0,12	0,42

Key: EECA = East Europe and Central Asia; LAC = Latin America and Caribbean Countries; MENA = Middle East and North Africa; SEAO=Southeast Asia and Oceania; SSA = Sub-Saharan Africa

Source: Licker *et al* (2010).

FIGURE 20 Trend of global yield change of major cereals.

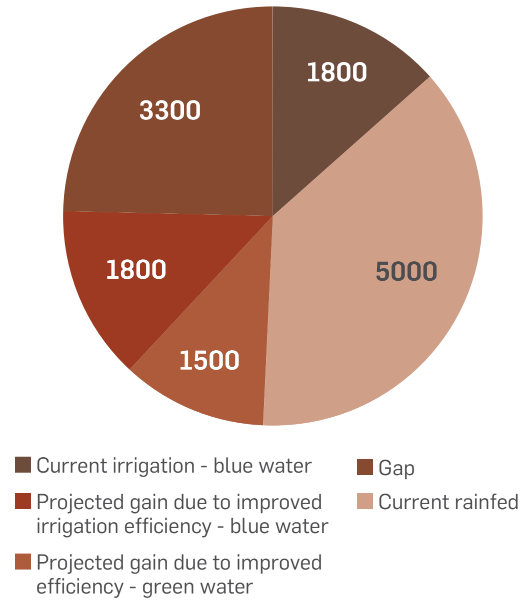


Source: Computed from Bruinsma (2009).

A higher water use efficiency is required

The increased demand for water calls for the improvement of water use efficiency to minimize or close the large gap between water supply and demand in the future under land use and water use efficiency (BAU). Under BAU and without use of water for biofuel production, water demand in 2050 will exceed water supply by 3,300 km^3 (Figure 21). Additional water withdrawal of 5,600 km^3/year is required to eliminate hunger and undernourishment and to feed the additional three billion inhabitants in year 2050 (Falkenmark, and Rockström 2004). This means almost doubling the current withdrawal of 7,130 km^3 2050 (CA 2007). Decreasing water supply is also a result of many types of land degradation (deforestation and land clearing, crusting, etc). This affects storage and availability of green water (soil moisture), which in turn reduces terrestrial ecosystems capacity to provide biomass and regulating services – such as carbon sequestration (Rockström *et al* 2009). This means an integrated approach is required to close the gap.

FIGURE 21 Current and future (2050) water use (km^3/year).



Source: Calculated from Shiklomanov (2000).

To achieve the required growth crop yield in low income countries and to address the increasing water shortage problem, more efficient water use is required. The average water use efficiency in rainfed systems in the arid and semi-arid areas in Africa is about 5,000 m^3 of water per ton of grain, but if supplemental irrigation of only 100 mm per year is used, crop yield doubles and reduces the water use to 2,000 m^3 (SIWI *et al* 2005). At a global scale, improving rainfed water (green water) use efficiency could reduce the water demand by 1,500 km^3/year or 80% of the current irrigation water (Figure 21). Expansion and improvement of productivity of the irrigation systems has the capacity to increase 1,800 km^3 (Figure 21). This would still leave a gap of 3,300 km^3 between supply and demand. This means efforts to close the water demand-supply gap should pay due attention to green water – which despite having a large potential to contribute to water demand has received limited attention in water development and management. In addition to food, about 90% of the global green water is required to sustain ecosystems (Rockström 1999).

Biodiversity

The Convention of Biological Diversity (CDB) established a goal to protect at least 10% of ecological regions (CBD 2011). Of the 825 terrestrial ecoregions, regions with large number of species and distinct habitat types, 56%

report more than 10% of their protected (CBD 2011). As observed above, the rate of biodiversity loss is almost 10 times the safe earth boundary. Despite the alarming biodiversity loss, the increase in the protected area provides the hope of reducing this unsustainable trend. This will only be possible if the governance challenges discussed in section 3 are addressed.

Choices for managing land use for multiple objectives and critical areas of global coordination

Since land area is fixed, all types of land uses are competing for the same land. The choices have to be defined by the ultimate benefits of land use – human welfare. As argued throughout this paper, all ecosystem components have an intricate interrelationship with one another. For brevity, our discussion below focuses on a selected number of choices with significant and direct trade-offs only.

For example, the mitigation of climate change under UNFCCC, conservation of biodiversity under CBD and prevention of land degradation under UNCCD by their nature are interdependent and could be simultaneously achieved. A recent international workshop promoted a nexus approach in which programs on interdependent ecosystems – such as food security, water and energy – are planned and implemented in a synergistic way. An example of synergistic objective is the Niger's reforestation program – which has covered 5 million hectares of planted or protected trees – provides 0.5 million tons of grain per year and sequesters carbon (Beddington *et al* 2011). The trees also provided fuelwood, medicinal plants and improved soil fertility. The world is increasingly realizing this potential and international cooperation on environmental management and governance has increased in the past two decades (Biermann *et al* 2010).

Regarding food, bioenergy, biodiversity, and reduction of GHG emissions, the world has to balance the three objectives using integrated LUCC modeling to find solutions which maximize human welfare. Studies have cast doubt on the efficacy of biofuels as mechanisms for reducing GHG using current technologies. In an attempt to achieve this, the current EU biofuel sustainability criteria passed in 2009, requires that liquid bioenergy should lead to CHG emission reduction of at least 35% and gradually increasing to 60% and should not be produced from raw materials grown on land of high biodiversity value or carbon stock (Nillson and Persson 2012). Such mandate considers two objectives – GHG emission reduction and conservation of biodiversity and sets an example of multiple objectives mandates. However, the EU biofuel sustainability mandate does not directly address food security aspects and does not consider indirect LUCC (iLUCC) (CEC, 2010). Achieving the EU biofuel sustainability mandate is a challenge since it does not consider iLUCC and it requires a constantly updated biodiversity data in order to trace the impact of feedstock production on biodiversity. This could be a big challenge in developing countries – especially in countries with weak institutions.[12]

Second generation feedstock is being advocated to reduce the food-bioenergy trade-offs. Research is needed to develop the second generation feedstock and renewable energy with minimal or no competition with food production. For example Gates (2011) notes that US investment in renewable energy is low and increasing research investment will have long term payoffs.

The agriculture sector is often not given the political attention and commitment that it deserves, especially in developing countries where trends over the last two decades indicate reduced allocation of national development budgets to agriculture. Furthermore there has also been a substantial decline in multilateral lending and bilateral aid for this sector. This trend is contrary to the increasing global cooperation on environmental conservation. The two food price spikes in the past five years should be seen as a wake-up call for national governments and the international community to invest in agriculture. This will help close the wide potential-actual yield gap in developing regions and consequently reduce land conversion to agriculture. However, agriculture investments should be made with multiple choices - ensuring food security and environmental objectives are addressed.

Achieving food security also requires reducing post-harvest losses, which are high in both developing and developed countries. Post-harvest food losses could be reduced by investment in processing and storage investment in developing countries and by public awareness in developed countries. Reduction in post-harvest losses will enhance food security and reduce the demand for additional land, energy and other resources.

Prospects for international instruments for land use change management

International market conditions provide a great potential for ensuring sustainable land and water management practices for ecosystems. Recent development in the carbon market illustrates the increasing international cooperation. Until the mid-1990s, international carbon market was negligible (Mol 2012). However, the climate change mitigation efforts broke what Beck (2005) called the national-state container and international carbon markets have increased dramatically (Mol 2012). The success stories in Brazil, Indonesia discussed earlier demonstrate progress, which could be made if international support is given to a country with strong policies and strategies to address unsustainable land conversions. Daunting challenges remain on implementing various global environmental programs. But the increasing international cooperation in land and water management has increased significantly since the first Rio summit in 1992 – thanks to United Nations concerted efforts to promote international cooperation (Sanwal 2004). Participation of the private companies and voluntary carbon market (VCM) initiatives by environmentally conscious companies also offers some opportunities for improving carbon markets but as discussed below, market based strategies still face challenges in countries with poorly developed markets and institutions. Though VCM only accounted for 0.3% of carbon traded in 2010 (World Bank 2011a), it is increasing and offers an opportunity to expand if conducive environment is created to enhance participation.

However, implementation of PES has been expensive and in some cases hard due to the weak institutions in developing countries – where it is cheaper to pay for PES and where degradation of biodiversity is more severe. For example, Bruce *et al* (2010) observed weak land tenure systems in areas high carbon density. The fragile states could also fail to meet the REDD+ and other international requirements, hence limiting their applicability and effectiveness. This means implementing PES may need to incorporate capacity building of local and national governance structure in fragile states.

The prospects of climate change negotiations are not bright and the carbon market trend levelled off in 2009 and showed a slight decline in 2010. This puts into jeopardy the international cooperation on climate change and on other initiatives. Of concern are the uncertainties surrounding the compliance market and additionality. For example, most decisions and rules and regulations on the REDD+ funded by governments and international organizations are still pending.

As argued above, synergistic programs – providing several ecosystem services are more likely to have greater pay-off and be more sustainable than single-objective programs. This suggests the international cooperation on carbon and other ecosystem service initiatives need to explore synergies among national and international sustainable development conventions such as UNCCD, CBD, UNFCCC and others should explore closer collaboration to achieve synergistic their objectives, namely, combating land degradation, conservation of biodiversity and carbon sequestration. This is in line with the Agenda 21 spirit which promoted cooperation and building on synergies among ecosystem initiatives. A new approach is also called for to strengthen the economic incentives for sustainable land use on a strong evidence base. Such an approach following a cost of action versus cost of inaction approach regarding land and soil degradation could go a long way toward mobilizing public and private investment for sustainable land use. A related initiative on “Economics of land degradation” (ELD) has been started in 2011 by UNCCD, Germany and the European Commission (Nkonya *et al*. 2011).

Annex 1

Major LUCC models

LUCC models have grown in the past few decades along scientific discipline and theoretical lines (Verburg *et al* 2004), which – as it will be seen below – have tended to merge over time. A comprehensive review by Heinstermann *et al* (2006) categorized LUCC models into two major groups: geographic models – which are rooted on the natural science discipline and therefore focus on the supply side – and economic models – which are rooted on the social sciences and focus on the demand side. Yet, a third group of models – also based on natural sciences – have focused more on land use and its impact on ecological services (Verburg *et al* 2004), especially regulating and supporting services. For example, the species distribution models (SDMs) have been developed to determine the biogeography and other ecological aspects (see Franklin and Miller 2009 for a comprehensive review). Recent models have increasingly combined both natural and social sciences – to study impact of LUCC change. Economic models have taken specific direction on ecology (ecological economics) and have used geographic approaches to analyze LUCC. Similarly, geographical models have taken both economic and ecological directions to analyze LUCC (economic geography and ecological geography). We discuss the three modeling approaches and the integrated models, which combine more than one scientific discipline. The focus of discussion of the individual scientific discipline is on their original scientific approach.

Geographic models

The **geographic models** are spatially explicit and they analyze the drivers of LUCC change and how this is related to the land properties and its suitability for different types of use. Geographic models assume that prices and other international feedback variables exogenous (Schneider *et al* 2011). Due to their spatial focus, geographic LUCC models have greatly contributed to the development of geographic information systems (GIS). Several geographic LUCC models have been designed but a couple of examples illustrate their strengths and weaknesses:

i. *Empirical-statistical.* These assume that the current relationship between LUCC change and its drivers will remain the same in future and such relationship is developed using regression analysis – hence its name of empirical-statistical. An example of empirical-statistical models is the CLUE model (Veldkemp and Fresco 1996). CLUE assumes that the cause-effect relationship holds for only a set of sub-regions with homogeneous biophysical and socio-economic characteristics. The strength of the empirical-statistical approach is in its ability of exploiting historical data to predict future trends and patterns. Biophysical and socio-economic characteristics are overlaid to determine their correlations, interactions and the changing suitability patterns. However, the authors acknowledge that the regression approach used is unable to gain deeper understanding of the interaction of the drivers of land use change and its processes (Veldekemp and Fresco 1996). CLUE and other empirical statistical models are unsuitable for long-term LUCC projections – especially under circumstances which the historical trend is different from the future patterns. An empirical example of relevance to this point is that of Brazil – in which the past patterns of frontier expansion in to forested areas will likely not hold in the future, given that more of the production increases are coming from intensification on existing areas (Nassar *et al* 2011). Likewise, climate change and other global changes make prediction of future scenarios using empirical-statistical models less accurate, unless they are linked with projections of key economic and environmental variables that come from macro models. An example of this is the projection of land use change impacts on the Brazilian Amazon done by Nelson and Robertson (2008), based on a econometrically-based model of land use choice that is interacted with projections of agricultural prices from the IMPACT model. Although the conversion of forest land might differ from models that link the economic market modeling more explicitly with the land use simulations, and allow for two-way interactions between the two, the forecast accuracy is better than which would result in the absence of any market price projections.

ii. *Rule-based/process based.* Designed by Stephenne and Lambin (2001) and used in the Sudano-Sahelian region in West Africa, the rule-based/process based model – SALU (Simulation model of land use change) captures agricultural intensification once a threshold is reached. For example, nomadic pastoralism becomes sedentary livestock production as a result of conversion of grazing lands into crop production. Likewise, fallowing period is shortened or eliminated as a result of increasing population density. The SALU and other related models combine spatially explicit data with qualitative reasoning (fuzzy logic). The appeal of SALU is its combination of qualitative and quantitative approaches and use of the threshold and other rules. However, use of qualitative reasoning could complicate its application to a larger area with diverse socio-economic characteristics.

Economic models

Economic models use welfare optimization principle either explicitly or implicitly (in terms of reduced-form equilibrium relationships that represent first-order necessary conditions of optimization) in order to model production and consumption behavior for agricultural and non-agricultural goods. The usage of various productive factors – such as land – can then be linked to the market equilibrium-based outcomes either implicitly (by dividing production by yield, for example) or explicitly, in those cases where land requirements per unit production are directly modeled with an input–output relationships or in terms of an explicit upper-bound constraint on available land resources that limits the expansion of area response functions for agriculture. Production relationship, consumer preferences are parameterized and fitted in the simulation model (Verburg *et al* 2004), and then drivers of consumption and production change can be represented by exogenous drivers (such as population, urbanization, technical progress or climatic conditions, among others) and can be varied according to alternative scenarios. The simulated outcomes give rise to land use changes that can be constrained or allowed to adjust according to rule- or market- based mechanisms A number of economic approaches to modeling land use change apply econometric analysis to historical data in order to establish the relationship between LUCC and its drivers (Ibid). Other economic LUCC models have used theory and biophysical science laws to establish the LUCC-driver relationship while others have used expert knowledge. The expert opinion approach always uses cellular automata approach, in which an expert defines the interaction between land use at a certain location, the land use type and surrounding conditions of a local area (Ibid).

The LUCC economic models involving regional or global models use different extensions of the von Thünen land rent theory[13] and Ricardian model of international trade – which in its simplest form – has two goods and labor as the only factor of production. International trade is dictated by comparative advantage of the trading countries. One country trades one product – in which it has comparative advantage (based on technology) to produce – to another country (Feenstra 2003). Additionally, economic models typically use elasticities of supply and demand to determine the response of production and consumption to price changes, so that a market equilibrium for agricultural and non-agricultural goods can be determined. Where area response is modeled separately from yield response – the amount of available land can then be imposed on these models as an exogenous constraint. Some economic models will allow the land allocations between sectors to remain fixed – such that agricultural land remains constant, although it can be allocated differently across different crops – whereas others might explicitly model how land might be allocated differently across agricultural and non-agricultural sectors, according to relative levels of economic return. In either case, prices are used to determine the allocation of land across different economic activities such that demand and supply of land-based products and services are determined by a market-mediated equilibrium and subject to limits on land usage that are either imposed as exogenous constraints or allowed to shift endogenously to the equilibrium solution. The economic models that are able to simulate LUCC always treat land as a factor of production – although the market for that factor may or may not be modeled explicitly. Their focus tends to be on modeling the outcome of land use rather than on the land allocation mechanism, itself – although this varies across economic models. The LUCC economic models are in two major categories: The partial equilibrium (PEM) models and computable general equilibrium (CGE) models. Recent attempts to combine PEM and CGE models have also been made, and will be given some discussion in what follows.

i. *Partial equilibrium models (PEM)*. These models take into account only some sectors (e.g. agriculture) and do not model the feedback from other sectors – especially as concerns the employment of productive factors such as labor and capital. Considering only a limited number of sectors allows PEM models to do an in-depth analysis of such sectors and focus on features of particular interest – such as yield response to environmental factors, or the inclusion of details of drivers of LUCC and other micro-level details that help to improve their prediction accuracy. Examples of PEMs are the IMPACT (International Model of Policy Analysis and Agicultural Commodty Trade' (IMPACT) model (Rosegrant *et al* 2001; 2009), which focuses on the agricultural sector; the agLU model (Sands and Leimbach, 2003) and FASOM (McCar 2004), which model interactions between agriculture and forest. Weaknesses of PEM are inclusion of only primary products or first stage processing products – excluding processed products, whose importance in world market is increasing. By their nature, the PEMs also ignore feedback from other sectors and this limits their ability to take into account the important implications on land allocation decisions made across sectors. For example, the IMPACT model does not take into account the impact of agricultural land expansion on forest products supply and demand. Neither does it consider the impacts on biodiversity and other ecological goods and services – which is common in almost all economic models.

ii. *General equilibrium models.* CGE models take an economy-wide approach to modeling economic market phenomena, and make a closer link between production and consumption activities by allowing the income from economic production to flow back to consumer households in the form of wage payments for labor or rental payments to factors owned by households such as land or capital. These models, while analyzing the particulars of one sector of interest, also take into account its interaction with all other sectors in the economy in terms of the competition that might exist for scarce factors such as land, capital and labor. All of these sectors (markets) are assumed to be in equilibrium at the same time, and their relative economic productivities and profitability determine their ability to attract or 'bid away' resources from each other. While most CGE models were developed to look at the impacts of trade and other government policies on consumer and producer welfare, they have been expanded in more recent applications to look more closely at resource allocation issues such as land. Examples of CGE models, which have been modified to analyze LUCC include variants that have been developed within the Global Trade Analysis Project (GTAP) framework. The family of GTAP-based CGE models include GTAP-E (Burniaux and Truong, 2002), which was expanded to consider energy issues in more detail, and has been extended for a number of analyses that look at the impact of biofuels on agricultural and non-agricultural sectors (Hertel *et al* 2008). Another extension of the GTAP model looks at LUCC more explicitly through the inclusion of land classifications based on the IIASA-FAO characterization of agro-ecological zones (FAO and IIASA 2000; Fischer *et al* 2002), which allows the productivity of agricultural and non-agricultural activities to be differentiated across different land classes. This variant is called GTAP-AEZ (Lee 2005; Lee *et al* 2005) and has been used more recently in the analysis of land use impacts of biofuels expansion (Golub *et al* 2008, 2009), and contains a mechanism for analyzing the trade-off between agriculture, forestry, pasture and other land uses. A number of attempts have been made to include features from models with more details on biophysical process into the GTAP family of models, in order to better represent the interactions of agriculture (and non-agricultural sectors) with land use. One example is the GTAPEM variant (Hsin *et al* (2004), in which the integrated assessment model IMAGE (Bouwman *et al* 2006)was used to obtain crop yield responses and requirements for animal feed, that were calculated consistently with projections of production quantities of crop and livestock commodities coming from the extended agricultural sector of the GTAPEM model. GTAPEM only considers agricultural land us, and allows for substitution between primary and intermediate products (Heistermann *et al* 2006). The GTAP-AEZ variant, by contrast, takes the land use requirements for forest and livestock and allow them to compete with the land used for agriculture. Additionally, the fact that CGE models focus on the flows of revenues and payments within the national and global economies means that, in many cases, they measure land usage in terms of its share in the total value of production, rather than its explicit physical area. Despite their broader approach however, many CGE models do not take into account intensification such as increasing fertilizer use in order to address lower yields on degraded lands (Lambin *et al* 2001) – although this has been improved in order to address the complicated 'indirect' land use issues related to biofuels expansion (Hertel *et al* 2010). Additionally, positive and negative externalities of production are rarely taken into account by many of the models that simulate land use change. For example, impact of LUCC change on GHG emission is taken into account by integrated assessment models with focus on climate change — such as the GCAM (Edmonds *et al* 1997, Wise *et al* 2009a, 2009b), IMAGE (Bouwman *et al* 2006) and AIM (Matsuoka *et al* 2001) models to name a few – and only a subset of these models (like the IMAGE model) have LUCC sub-components that enable them to take these externalities into account, especially in terms of their impact on yields. Only few PEM and CGE are fully dynamic – i.e., all equilibria at any given time are solved simultaneously – assuming the economic agents have perfect knowledge of the future. Examples of fully dynamic models include G-cubed (CGE) and FASOM (PEM) (Ibid) and WATSIM (Kuhn 2003). GTAP-AEZ and GTAPEM are typically solved within a static framework and do not take into account the changes that occur over time in the relationship between drivers of LUCC and their impact on land productivity (Heistermann *et al* 2006).

iii. *Integrated PEM and CGE economic models:* To take advantage of strength of CGE and PEM modeling discussed above, attempts have been made to combine PEM and CGE modeling. For example Blitz and Hertel (2011) used a PEM (common agricultural policy regional impact – CAPRI) and CGE (GTAP) model to analyze the impact of biofuels mandates in the European Union and US on direct land use change (LUC) and indirect LUC (iLUC). The combined PEM-CGE allowed them to estimate both global and detailed regional LUCC changes and nutrient use.

Integrated geographic and economic LUCC models

A combination of geographic and economic models has emerged as an attempt to overcome the weaknesses of the two groups discussed above. Verbug et al (2008) observe that LUCC models, which integrate several disciplines, are better predictors of future LUCC than those which focus on only one discipline. While most of integrated geographic and economic LUCC models have simply combined the economic and geographic models, few others have been designed from scratch reflecting both economic and geographic characteristics. For example, the FARM model (Darwin *et al*., 1996) was originally a CGE model, which has integrated spatially explicit geographic modeling based on classification similar to those used in the GTAP-EAZ variant. Likewise, LEITAP (van Meijl *et al* 2006, Banse *et al* 2008) is a variant of the GTAP model in which the tradeoffs between agricultural and non-agricultural land uses are modeled with inputs from the integrated assessment model IMAGE – such that the total availability of land and its changes in productivity are taken from the IMAGE model outputs. This combination of economic and geographic model features takes advantage of the strengths of each modeling approach (Heistermann *et al* 2006). A number of economic models have tried to combine the geographic specificity of agronomic models that characterize yield potential and response with the higher-level representation of market-based equilibrium for agricultural supply, demand and trade. The GLOBIOM model of IIASA (Sauer *et al* 2010, Havlik *et al* 2011) takes the pixel-level simulations of crop and vegetation production from the EPIC model (Williams 1995), and combines it with aggregate-level representation of consumption and trade. The MAgPIE model (Lotze-Campen et al 2010, Popp *et al* 2011) uses the LPjML model (Bondeau *et al* 2007) to simulate pixel-level yield response to soil quality and water availability with an optimization-based allocation of land, in order to meet defined targets of food consumption across various regions in the future. Both the GLOBIOM and MAgPIE models integrate the agronomic modeling within the simulation structure more closely than is done with the, IMPACT model, in which the pixel level calculations of potential crop yield are only used as an adjustment factor to project how the more macro-level yield relationships of the model will shift under different outcomes of climate , and are carried out as separate calculations outside of the market equilibrium simulations (Nelson *et al* 2009). The results from the crop simulation are then aggregated to compute the national and regional level impact of both biophysical and socio-economic drivers of LUCC. There is still room for exploiting the potential of the integrated models to overcome the weaknesses of each. A lot remains to be done to take into account the complex LUCC systems and the consequent disciplinary approaches (Verburg *et al* 2004).

Ecological and climate change models:

Ecological models have increasingly been used to help planning of nature conservation programs and recently impact of LUCC on climate change (Brown *et al* 2002). Exploration in the study of spatial distribution of species and their habitats and endemic (specific to a geographical location) species has especially increased in an attempt to understand the alarming species extinction rate and the required conservation strategies. Similar to the geographic and economic models, species distribution modeling (SDM) has taken advantage of the improved spatial techniques and availability of satellite data (Franklin and Miller 2009). SDMs major goal is to determine LUCC, such that anthropogenic ecosystems have minimal alteration of the ecological conditions required for maintaining the species distribution. The SDMs have been developed along three approaches – (i) species-environment correlations (ii) expert opinion – and (iii) spatially explicitly and statistical and empirical models (Guisan, and Thuiller 2008). Species area relationship (SAR) models are a special form of SDM with focus on species extinction. SAR seeks to determine the conservation value of an ecoregion by the number of species in area relative to its potential species abundance, which is modeled using the simple power-law:

$$S = cA^z$$

Where S = species richness, A = land area, and c and z are constants (Lee & Jetz, 2008; Giam *et al* 2011). Giam *et al* 2011 used the historical trends of species abundance and drivers of biodiversity loss and spatial human population trends to predict future trends of species richness and to identify priority areas for conservation. Such analysis does not consider other socio-economic drivers of biodiversity loss and this further emphasizes the need for integrated models discussed above and below.

Recently, ecological footprint models have also attempted to determine the biocapacity and ecological footprints (FGN 2010; WBGU 2011). These models determine the biocapacity and the population's consumption of provisioning services and regulation services. Ecological economic models – discussed in detail below – have also used traditionally economic models to allocate land to biodiversity and other conventional ecosystems. Using IMAGE 2.2 LUCC model and taking climate change into account using IPCC SRES scenarios, van Vuuren and Bouwman (2005) observed that the ecological footprint (EF) in 17 regions of the world and determined that EF to increase from the 5.6 global hectares (Gha) in the baseline year to 6.2 - 8 Gha in 2050 due to increasing income, population and changing food tastes and preferences.

Though there has been significant development in the predictive modelling of biogeography, evolution, conservation biology and climate change, the link between SDMs and ecological theory has remained weak (Elith and Leathwick 2009). Strengthening this link will help to understand biotic and abiotic interaction. Like the economic models, SDM projections are based on the assumption that the historical trend and pattern of species holds in the future, i.e., no evolutionary adaptation and no global dispersal from international trade (Dormann 2007).Just as is the case with the economic models, SDM modeling has not been able to capture some drivers of species distribution and abundance and Keith et al., 2008 and others recommend using multiple models to reflect the feedback among potential habitat shifts, landscape caused by land use patterns, landscape patterning caused by altered disturbance regimes), and demography for a range of species functional groups is a way to develop guidelines for assigning degrees of threat to species. We discuss below the ecological economic models as examples of integrated ecological and economic models.

Ecological economic models have combined ecology and economic principles to determine land allocation to biodiversity (wilderness), agriculture, forests and other anthropogenic ecosystems. For example Read (1997) sought to allocate non-forest and non-barren land to land wilderness (for biodiversity benefits), agriculture, carbon sequestration forests, and biofuel production by solving for an optimal solution that maximizes global welfare. Considering only studies which included marketed and non-marketed goods and services, Balmfold et al (2002) reviewed 300 studies to determine the losses of ecosystem services resulting from conversion of conventional (natural) ecosystems to anthropogenic ecosystems (agriculture, planted forests, rangelands, and settlements). Their review concluded that loss of non-market services was greater than the marketed marginal benefit of conversion. In the four biomes considered, the mean loss in total economic value (TEV) due to conversion was about 55%±13.4% compared to the TEV of the natural habitat. They further give a caveat that their results do not mean that there are no conversions which are beneficial but rather, their results suggest that present conversions of the remaining natural habitats is not likely to be beneficial for global sustainability – if value of all ecosystem services are taken into account. Their study further shows that the benefit-cost ratio of conservation of natural habitat was 100:1. Such conclusion have been drawn by other recent studies, which suggest that sustainable development is feasible only through increased agricultural productivity rather than conversion of virgin lands.

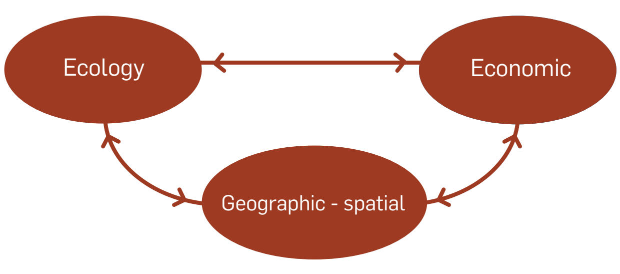
Integrated economic-geographic-ecological models.

As has been argued throughout, integrated economic, geographic and ecological models are required to capture the multiple drivers of LUCC and objectives of ecosystems. Such an approach has been taken by few studies. For example, in a team involving economists, ecologists and geographers, Pfaff *et al* (2000) proposes integrated method for analyzing LUCC change using ecological, economic and geographical models. SDMs are rarely used in planning conversion of land to agriculture (Evans *et al* 2011) or other anthropogenic ecosystems. Evans *et al* (2011) used species distribution models (SDM) to allocate land to biofuel production in US such that biofuel production had minimum impact on biodiversity. Such consideration is important in developing models which are environmental-smart. The MA (2005) also analyzed the LUCC and its effect on ecosystem services. The MA (2005) also examined the social impact of LUCC and demonstrated a fairly comprehensive approach to analyzing the ecosystem services. For models to fully account for the total economic value of LUCC and trade-offs, they should be integrative, multidisciplinary to the extent that they take into account changes in all terrestrial ecosystem services. The Patuxent Landscape model (Voinov *et al* 1999 and Geoghegan *et al*., 1997) and General Ecosystem model (Fritz *et al* 1996) are examples of the integrated models. The Patuxent Landscape model simulates ecological and hydrological conditions using ecological modules and uses economic modules to account for the land use changes. In the General Ecosystem module, the economic modules and ecological modules are coupled such that results of each component are used interchangeably to provide feedback mechanisms in each module. For example, LUCC results from the economic model are used as inputs into the ecological modules while ecological outputs (e.g. water depth, habitat health, etc) are used as inputs into the LUCC economic models.

Nilsson and Persson (2012) considered global governance of three interacting earth's systems: climate change, biodiversity, freshwater and land use. They conclude that earth's four systems can be governed to ensure that the planetary boundaries are not reached. But such governance requires elaborate political and institutional arrangement,

Figure 22 presents the schematic framework of the integrated models required to reflect the full value of the ecosystem services. Unfortunately, economic models – largely based on market clearing principles fail accommodate externalities of land use activities (Sukhdev 2008). It is also hard to assign an economic value to biodiversity and regulating services. The tranboundary nature of benefits and costs of land use and management also complicates economic modeling. A country which seeks to address food security could achieve its objective by clearing rainforests, which have global carbon-sequestering benefits. For example, we saw above that DRC has a strong incentive of clearing the Congo rainforest to address its food security. To achieve the global objective of carbon sequestration and other forest services, some compensation mechanisms – such as PES discussed earlier – are required to incentivize land users not to clear forests. Integrated modeling including economic, geographic and ecological models is required to determine the spatial distribution of benefits and costs of LUCC and land management practices. Such models will also help to determine trade-offs of different LUCC. For example, Rudsepp-Hearne *et al* (2010) observed that intensively managed ecosystems (crops & pork production) were negatively correlated with regulating services (carbon sequestration, soil organic matter and soil phosphorus retention) and cultural services (tourism, forest recreation, etc). However, the same study observed strong positive correlation of natural ecosystems (regulating and cultural services).

FIGURE 22 Integrated LUCC models.



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Endnotes

1 We follow the FAO definition of forest – a land mass with at least 10% of its area covered by trees.

2 The goals are: 1. Promote the conservation of the biological diversity of ecosystems, habitats and biomes; 2. Promote the conservation of species diversity; 3. Promote the conservation of genetic diversity; 4. Promote sustainable use and consumption; 5. Pressures from habitat loss, land use change and degradation, and unsustainable water use, reduced; 6. Control threats from invasive alien species; 7. Address challenges to biodiversity from climate change, and pollution; 8. Maintain capacity of ecosystems to deliver goods and services and support livelihoods; 9. Maintain sociocultural diversity of indigenous and local communities; 10. Ensure the fair and equitable sharing of benefits arising out of the use of genetic resources; 11. Parties have improved financial, human, scientific, technical & technological capacity to implement the Convention.

3 Brazil and the Congo are respectively the first and second countries with largest tropical forest area.

4 After Nigeria and Ethiopia, with populations of 160 million and 84 million people respectively (FAOSTAT 2011).

5 See ClientEarth (2009) for an analysis of WTO jurisprudence about trade of natural resources.

6 www.theredddesk.org/activity/memorandum_of_understanding_on_environmental_cooperation_between_the_state_of_acre_of_the_f

7 http://news.mongabay.com/2009/0915-cerrado.html

8 LUCC was an International geosphere-biosphere program (IGBP) and international human dimensions of global environmental change program (IHDP) project, which organized a workshop to discuss land use and land cover change (Veldkamp and Lambin, 2001).

9 Ehlirch and Ehlirch (2009) revisited the "population bomb" publication and argued that its main message is still valid, though they admit the exaggeration resulting from the sensationalist title.

10 Following Brazil's commitment at the climate change summit in 2008 to reduce deforestation in the Amazon to 20% of its rate in 1996-2005, Norway committed US$1 billion to support achievement of this target. The Brazilian government also initiated a forest moratorium, by paying ranchers and soy bean farmers who do not cut the forest. The protected area of the Amazon was also increased (Nepstadt *et al* 2010).

11 Formed in 2011 as part of the UNFCCC COP-17 to synthesize empirical evidence into policy actions for achieving global food security given the climate change (Beddington *et al* 2011).

12 See further discussion on international instruments below .

13 Land rent theory assumes that a land parcel is allocated such that it earns the highest possible rent given its attributes and location.